Analysis of Monitoring Data from Platte Lake, Michigan

prepared for

Michigan Department of Natural Resources

by

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Introduction

Platte Lake is a 10.6 km^2 natural impoundment located on the Platte River in Benzie County, Michigan (Figure 1). It has a mean depth of 7.7 meters, maximum depth of 29 meters, and water residence time of 5.9 months. Its 415 km^2 watershed was approximately 93% undeveloped in the early 1980's (MDNR, 1983). It has been classified as "oligo-mesotrophic" (MDNR, 1983), with relatively low nutrient concentrations, low algal productivity, and depletion of dissolved oxygen in bottom waters during the summer. Inflows from the Platte River are dominated by relatively constant groundwater discharges with high alkalinity. Seasonal decreases in water transparency occur as a result of calcium carbonate precipitation, which occurs naturally in similar hardwater lakes.

Since the 1920's, the State of Michigan has operated a fish hatchery on the Platte River approximately 14 km upstream of the Lake. In the early 1970's, the hatchery was expanded and production shifted from rainbow trout to salmon and other anadromous fish. Legal proceedings were initiated in the 1980's because of concerns about the impacts of discharges from the expanded hatchery on nutrient concentrations, water transparency, and related water-quality conditions. As a result of these proceedings, an 8-ppb total phosphorus goal was established by Court (Brown, 1988ab) and the State was ordered to alter the operation of the hatchery in a manner consistent with achieving this goal. With implementation of control measures, phosphorus loads from hatchery decreased from ~33% of the total lake load in 1976-1981 to <4% in 1993-1996.

Under the direction of a Court Master, intensive lake and watershed surveys were initiated to evaluate lake conditions relative to the goal, to quantify phosphorus sources to the lake, and to identify and evaluate potential control measures (Fuss, 1989). This report analyzes data collected under these surveys between 1990 and 1996. Less intensive monitoring data from 1976 to 1989 provide an historical frame of reference. Long-term responses to historical reductions in hatchery phosphorus load are quantified through data analysis and modeling. Current and projected future conditions are evaluated with particular reference to the 8-ppb goal and to a 10-foot transparency benchmark identified in legal proceedings.

Main chapters of the report describe data sources, watershed data analyses, lake data analyses, mass-balance modeling, discussions of current and projected lake conditions, recommended management strategies, and conclusions. Supporting data sets, data displays, and calculations are presented in Appendices A-G.

Data Sources

Appendix A summarizes monthly climatologic data that have been obtained from Internet sites maintained by the National Oceanographic and Atmospheric Administration (NOAA). Compiled data include precipitation, air temperature, sunshine, and wind speed. Air temperature and precipitation are derived from a long-term drought-analysis data set (1897-1997) compiled by NOAA for this portion of Michigan. Daily wind speeds from the Traverse City airport have been summarized as average speeds and average cubed speeds. The latter are approximately proportional to mixing energy input to water surfaces (Fischer et al., 1979). Sunshine data are from Sault Ste. Marie (closest station with this type of information).

Three data sets describing current and historical water quality conditions in Platte Lake and its watershed have been compiled to support this analysis:

1. **Court Master (CM) Data**. Intensive lake and tributary monitoring conducted under the direction of Mr. Joseph Fuss of Aquatic Systems Engineering (1990-1996); derived from three Excel files ("LAKE.XLS", "RIVER.XLS", & "HATCHERY.XLS").

2. **Historical Data**. Intermittent lake and watershed monitoring conducted by state agencies and contractors (1976-1989), extracted from files provided by MDNR (Bednarz, 1993); sources include STORET and reports by Grant (1979), Kenaga & Evans(1982), MDNR(1983), and 1987-1989 surveys by Evans (1987-1989, unpublished);

3. **Lay Monitoring Data**. Weekly Secchi depth readings collected by local citizens between May and September, 1977-1996; extracted from the "LAKE.XLS" file provided by the CM and from a file supplied by MDNR (Bednarz,1993).

Watershed flows and phosphorus loads computed using concentration and flow data extracted from the above files are summarized in Appendix B. The U.S. Geological Survey (USGS) has provided daily flow data from two regional gauges: Platte River near Honor (Station 04126740, 1990-1996) and Jordan River near East Jordan (Station 0427800, 1975-1996). Estimates of Platte River daily flows prior to 1990 are developed based upon correlation with Jordan River flow.

Lake monitoring data extracted from the above files are summarized and displayed in Appendices C-G. The CM's lake survey design and resulting data set are characterized by the following:

Location:	West Basin, deepest point in lake (~90 ft)
Duration:	~ 7 years, Nov 1989 - December 1996
Variables:	Total P, Transparency, Temperature, Dissolved Oxygen, Secchi, Chlorophyll-a, Alkalinity, pH, Lake Elevation
Sampling Frequency:	~biweekly; weekly in summer >96 Total Dates = 181
Sample Depths:	Surface,15, 30, 45, 60, 75, 90 ft 7.5-ft sample added after Feb 1993

	Total Dates x Depths = 1376
Replicates:	3 for each Date & Depth (Total P only)
	Total Samples = 4107

Alkalinity and pH were measured after August 1992. Phosphorus concentrations were reported to the nearest 0.1 ppb. Secchi depths on each date were reported to the nearest 0.5 foot. Chlorophyll-a concentrations were reported to the nearest .01 ppb. A single chlorophyll-a value was reported for each date, derived from a depth-integrated sample over twice the Secchi depth (~photic zone). The chlorophyll-a analytical procedure was changed from spectrophometric to fluorometric in 1993.

Triplicate measurements of total phosphorus were reported in source data files for all lake, watershed, and hatchery sites monitored in 1990-1996. Source files do not indicate whether these represent triplicate samples or triplicate analyses on the same sample. The Court-Master's data-reduction algorithm involved deleting measurements with phosphorus concentrations exceeding the triplicate mean by more than 15%. This procedure was apparently designed to filter out unrepresentative values (outliers) which tend to occur in these types of data sets, even when extreme care is taken in collecting and analyzing samples. The 15% criterion is based upon a comment by Dr. Effler of Upstate Freshwater Institute: "other laboratories that perform low-level phosphorus analysis consider a 15% variation in readings as normal" (Fuss, Platte River Watershed Phosphorus Monitoring Program, Report No. 1, undated). The criterion has apparently not been justified based upon a systematic analysis of replication error using the specific sampling and analytical methods employed in the current monitoring program. The screening algorithm could introduce bias because it selectively deletes high values and does not allow for the possibility of outliers on the low side. For these reasons, an alternative data-reduction procedure is used in the following analyses. Except where supported by independent documentation of sample contamination in the source data files, deletion flags have been removed from all phosphorus observations in source data files. The median value is taken as a representative value for each set of replicates. This procedure has been taken from PROFILE, software developed for the U.S. Army Corps of Engineers to facilitate analysis and reduction of reservoir monitoring data (Walker, 1996). Taking the median of all observations has the effect of filtering out high or low outliers and is considered more robust and objective than taking the mean of screened observations. As shown below, computed annual-mean lake phosphorus concentrations are relatively insensitive to differences in data-reduction procedures.

Watershed Data Analysis

Flows & Loads at Monitored Sites

Phosphorus loads have been estimated from sample concentrations and daily flows at the following locations:

Platte River at USGS Gauge near Honor
North Branch Platte River at Dead Stream Road
Brundage Creek
Platte River Hatchery

Station locations are indicated in Figure 1. The following sections describe data and methods used to compute loads, flow and concentration dynamics at each site, and integration of results to estimate total watershed loads. Results are summarized on monthly and yearly time scales in Appendix B.

Phosphorus concentrations used in computing 1990-1996 phosphorus loads at the three watershed stations have been extracted from RIVER.XLS and HATCHERY.XLS. River sampling frequencies varied from weekly to monthly, with a data gap between June 1994 and February 1996. Brundage Creek was sampled consistently and more intensively because it was used in tracking hatchery phosphorus loads. Historical data (1980-1981, 1987-1988) from watershed sites have been extracted from files supplied by MDNR. These data are of limited use for calculating annual phosphorus loads because of the low sampling frequency (monthly) and lack of a continuous flow gauge in the watershed prior to 1990. Daily flows have been measured at the Platte River site by the U.S. Geological Survey (USGS, station 04126740) since 1990. Daily flows at the other watershed sites have been estimated based upon correlations with Platte River flows, as described below.

The FLUX program (Walker, 1996) has been used to calculate loads at the Brundage Creek, USGS gauge, and North Branch sites. FLUX integrates sampled concentrations over the entire daily flow record, while accounting for correlations among concentration, river discharge, and season. Daily sample concentrations have been represented by the median of triplicate values reported for each sampling date. Hatchery net phosphorus loads computed by the Court Master have been extracted from HATCHERY.XLS.

The sample concentration record at the USGS and North Branch stations extends from February 1990 to May 1994 and from March 1996 to December 1996. The elimination of sampling on the Platte River upstream of the hatchery after 1990 and the gap in the remaining stations between June 1994 and February 1996 are significant deficiencies in the monitoring program design. Direct monitoring of lake inflows is critical to formulating lake nutrient balances and developing a fundamental understanding of factors driving variations in lake water quality. Monitoring of lake inflows is normally given a high priority in lake diagnostic studies. As demonstrated below, a limited focus on hatchery phosphorus loads provides little insight into lake responses in recent years because they account for a very small portion (<4%) of the total load from other gauged and ungauged sources. Greater emphases on monitoring lake inflows and formulating overall phosphorus balances must be placed if the program is to be successful in establishing a predictive linkage between lake water quality and watershed inputs and a rational basis for management decisions.

The following table summarizes average flows and loads at watershed gauging sites for the 1990-1996 period:

Location	Flow 10^6 m^3/yr	Load kg/yr	Conc. ppb	Error C.V.
Platte R. @ USGS Station (excluding Hatchery load)	121.1	1565	12.9	4.0%
North Branch	31.3	497	15.9	3.4%
Brundage Creek	12.8	157	12.3	3.5%
Hatchery 1990-1996	-	190	-	-
Hatchery 1990-1992	-	316	-	-
Hatchery 1993-1996	-	96	-	-

Error coefficients of variation (C.V.'s) reflect uncertainty in the average load and flow-weighted-mean concentration at each station. Results indicate a relatively high degree of precision in the average load estimates. The standard error of the combined load from the Platte River above the USGS gauge and North Branch is ~70 kg/yr. Because of data gaps, limited number of high-flow samples, and potential drifts in the flow/concentration relationship over time, there is much greater uncertainty in estimates of watershed loads and concentrations for individual years. With improvements in operation, hatchery loads decreased from 316 kg/yr in 1990-1992 to 96 kg/yr in 1993-1996. Flow and concentration dynamics at each monitored site are described below.

Platte River @ USGS Gauge

The USGS site is located on the Platte River ~7 miles downstream of the hatchery and ~2 miles upstream of the Lake (Figure 1). The flow gauge has been operated since 1990 and represents the only continuous flow gauge in the watershed above the Lake. Figure 2 shows the daily flow hydrograph for 1990-1996. Symbols indicate dates of sampling events for the USGS and North Branch sites. Peak flows were consistently sampled in 1990, but usually missed in later years. Load estimates derived from FLUX account for correlations among concentration, flow, and season. Reasonable load estimates for the unsampled peaks and gap in the later portion of the record can be derived using data from the sampled peaks, assuming that the underlying flow/concentration relationship in the watershed did not change over the sampling period. A greater emphasis on sampling high flows would increase the accuracy and precision of future load estimates at all sites, particularly if estimates on monthly or yearly time scales are desired.

As shown in Figure 2, daily flows dropped below the range of sampled flows in 1995 and 1996. During this period, the flow/concentration relationship developed from sampled dates is extrapolated beyond the calibration range. Because of this extrapolation and the absence of samples, there is greater uncertainty in the load estimates during this period, as compared with the reminder of the 1990-1996 period.

Measured phosphorus concentrations at the USGS gauge have been adjusted to account for hatchery phosphorus load on each sampling date prior to calculating yearly loads. Hatchery loads are assumed to reach the site without attenuation and are considered separately in formulating lake phosphorus balances. The measured stream concentration on each sampling date has been adjusted by subtracting the average load from the hatchery in the corresponding month from the measured daily stream load and dividing by the daily stream flow. The FLUX outlier-detection procedure identified one suspect observation (concentration = 44 ppb on September 6, 1990); this sample has been excluded from load calculations for this site. The flow-weighted-mean concentration in 1990-1996 was 12.9 ± 0.5 ppb (mean ± 1 standard error).

Figure 3 plots adjusted concentrations against date, month, and flow. Sample concentrations have been paired with mean daily flows reported by the USGS on the corresponding dates. River concentrations vary seasonally; values typically range from ~20 ppb in winter/spring to <10 ppb in late summer and fall. This pattern most likely reflects influences of spring runoff and phosphorus trapping in upstream lakes during the summer. Higher concentrations measured in 1990 (> 30 ppb) were associated with higher flows (> 200 cfs). The positive correlation between concentration and flow most likely reflects influence of surface runoff during periods of high total flow. When adjusted for variations in flow, average concentrations were relatively stable over the 1990-1996 period.

North Branch Platte River @ Dead Stream Road

The North Branch discharges into the Platte River ~0.2 miles upstream of the Lake (Figure 1). This site reflects outflows from the North Branch watershed and Little Platte Lake. Figure 4 plots sample concentrations against date, month, and flow. Different symbols are used to represent earlier (1990-1994) and later (1996) portions of the record, when significantly lower flows were measured. With the exception of one high-flow sample, concentrations in 1996 were similar to those measured in 1992-1994. Seasonal variations in stream phosphorus concentrations are evident;

concentrations typically range from 12-50 ppb in May-July to <12 ppb in December. This seasonal pattern is distinctly different from that observed on the Platte River (Figure 3). In a study of the St. Paul water supply (Walker, 1992; Walker et al., 1989), similar seasonal patterns were observed in watersheds containing high percentages of wetlands. North Branch phosphorus concentrations also increased with flow, particularly in 1996.

Load calculations require estimates of daily flow for the entire sampling period. Source data files provided measurements of instantaneous flow on dates of sample collection. Daily flows have been estimated by correlating the sampled flows with daily flows from the Platte River gauge (Appendix B), using data from 1990-1994. As shown in Figure 4, measured flows dropped significantly when the flow gauging methodology was apparently changed from a staff gauge in 1990-1994 to a weir equation in 1996. In 1990-1994, measured flows averaged 26% of Platte River flows on the corresponding dates. These results are consistent with the ratio of drainage areas (31.1/118 mi^2 = 0.26). In 1996, measured flows averaged only 13% of the Platte River flows. The shift in flows may be partially attributed to leaking flashboards in the flow gauging structure, as noted in the source data file. Flow measurements at this site are generally problematical because occasional backwater effects from Platte Lake. Given the significant inconsistency in flows for the two periods, flows from 1990-1994 (consistent with drainage areas) have been used to calibrate the regression model for predicting daily flows based upon Platte River flows. To the extent that the drop in flow in 1996 was real and not an artifact of leaking flashboards or other measurement errors, flows and loads from this watershed may be over-estimated. Further investigation of discrepancies in the North Branch flow record is recommended.

The FLUX outlier-detection procedure identified one suspect observation (concentration = 71 ppb on January 21, 1994); this sample has been excluded from load calculations. The flow-weighted-mean concentration at the North Branch site for the 1990-1996 period was 15.9 ± 0.6 ppb, ~25% higher than that estimated for the Platte River site (12.9 ± 0.5 ppb). As indicated in Figure 4, concentrations were generally lower in 1992-1996, as compared with 1990-1991. The apparent decrease is not explained by variations in flow. When samples from 1990-1991 are used to calibrate the flow/concentration relationship, the flow-weighted-mean concentration for the 1990-1996 period is 21.2 ± 1.1 ppb. When samples from 1992-1996 are used, the flow-weighted-mean is 14.1 ± 0.5 ppb (similar to Platte River concentration). The apparent 7.1 ppb reduction in concentration is equivalent to a 224 kg/yr reduction in load under 1990-1996 flow conditions. The latter reduction is similar in scale to the average hatchery phosphorus load over the 1990-1996 period (190 kg/yr). It is unclear whether the apparent reductions in concentration and load from the North Branch between 1990-1991 and 1992-1996 represent a long-term change.

Brundage Creek

Brundage Creek discharges into the Platte River ~9 miles upstream of the Lake (Figure 1). Figure 5 plots sample concentrations against date, month, and flow. Seasonality is weaker than that observed at other stations, but there is a strong positive correlation with flow indicative of a runoff effect. Decreasing concentrations over the 1990-1996 period are partially explained by decreasing basin flows.

Daily flows for Brundage Creek have been estimated from Platte River flows based upon drainage area ratio (12.5/118 = .106). The FLUX outlier-detection procedure identified one suspect observation (concentration = 102 ppb on August 1, 1990); this sample has been excluded from load calculations for this site. The flow-weighted-mean concentration at this site for the 1990-1996 period was 12.3 ± 0.4 ppb, slightly below that calculated for the Platte River site (12.9 ± 0.5 ppb). When adjusted for flow and seasonal variations, FLUX diagnostic procedures indicate that concentrations at this site decreased at an average rate of 2.2%/yr.

Platte River Hatchery

Monthly inflow and outflow volumes, loads, and concentrations from the Platte River Hatchery between 1990 are 1996 are plotted in Figure 6. Net loads and concentration increases are plotted in Figure 7. Net loads, computed based upon the difference between the outflow and inflow loads, decreased from 316 kg/yr in 1990-1992 to 96 kg/yr in 1993-1996. Net increases in concentration between the hatchery inflow and outflow were generally less than 10 ppb in the latter period. Negative net loads were reported in four months between 1994 and 1996. In these situations, the hatchery was apparently functioning as a treatment facility for runoff loads from the watershed.

In 1993-1996, the computation of net loads became increasingly dependent upon taking the difference between two measurements (outflow load - inflow load) of similar magnitude and limited precision. Uncertainty in recent load estimates was further amplified by the fact that only a portion of the hatchery inflow load was directly monitored. Between 1990 and 1996, the Brundage Creek intake accounted for 54% of the total hatchery inflow. The remainder of the inflow was taken from the Platte River. Direct monitoring of concentrations in the Platte River inflow was discontinued in 1990. The assumption that the Platte River inflow and Brundage Creek inflow concentrations are identical may not hold, particularly on a daily or weekly time scale. Computations in the source data file (HATCHERY.XLS) contain adjustments for variations in the Platte River inflow concentration based upon antecedent rainfall. The basis for these adjustments is unknown. Direct monitoring of inflows from the Platte River is recommended for the following reasons:

1. to provide more accurate and precise estimates of net hatchery loads;

2. to further establish a baseline for the watershed above the hatchery; and

3. to provide a basis for evaluating the fate of hatchery phosphorus loads in the river between the hatchery and the Lake.

Although lake mass balances formulated below assume that all of the hatchery phosphorus loads reach the lake without attenuation, this is not necessarily the case. Phosphorus balances developed by Grant (1979) and Kenaga & Evans (1982) assumed 10% attenuation. Potential mechanisms for load attenuation would include algal uptake and adsorption to calcite particles that may form in the river during summer, when river temperatures are high and pH is elevated. Temperatures above 21 deg-C and pH levels above 8.5 were recorded in the river below the hatchery in June-July of 1987 and 1988. These conditions would be conducive to calicite precipitation. Although phosphorus adsorbed to calicite particles may eventually reach the lake, a portion of the adsorbed phosphorus may not be biologically available. Further studies would be needed to evaluate this potential mechanism. Consistent monitoring of river concentrations above the hatchery and above the lake would be required to evaluate potential phosphorus transformations and transport losses in the river between the hatchery and lake. Given the magnitude of current hatchery loads, however, they may be difficult to detect at the lake inflow in the presence of background variability in river flows and concentrations.

Total Watershed Loads

Monthly flows and loads from the following subwatersheds have been calculated using results at the four monitored sites:

Upper Platte River = USGS Gauge - Brundage Creek - Hatchery
Lower Platte River = (USGS Gauge + North Branch - Hatchery) x 0.086
Watershed = Brundage + Upper + Lower + North Branch
Total, Including Hatchery = Watershed + Hatchery

Results are summarized by month and year in Appendix B. Lower Platte River flows and loads (below the USGS gauge, excluding North Branch) are estimated based upon drainage area ratio. These values do not reflect additional loads in urban runoff from developed areas around the lake, which are considered separately below in formulating lake phosphorus balances.

Monthly loads have been calculated using the FLUX time series interpolation algorithm. Estimates reflect the underlying relationship between concentration and flow/season calibrated to data from sampled dates. Drift in the relationship is reflected by interpolating errors over time. Estimates for unsampled months (June

1994-February 1996) are relatively uncertain because there is no adjustment for drift and daily flows dropped below the range of sampled flows (Figure 2).

Figure 8 shows monthly flows and loads from the total watershed in relation to the hatchery contributions. The relatively small variation in total flow reflects the apparent importance of groundwater discharge (base flow) as the primary water source in the basin. Hatchery loads averaged 11% of the total watershed loads in 1990-1992 and 5% of the total loads in 1993-1996.

Climatologic variations were largely responsible for higher watershed loads in 1990, as compared with the remaining years. Peak loads were measured in March and June of 1990 during periods of high runoff and/or precipitation. Figure 9 plots monthly watershed loads against precipitation and runoff. Runoff has been estimated from the Platte River daily flow time series using the following algorithm:

Base Flow = 30-Day Antecedent Minimum Daily Flow
Runoff = Daily Flow - Base Flow

Different symbols are used to distinguish months with relatively high load response (January - July) from months with relatively low response (August - December). Seasonal differences in load response may reflect seasonal variations in groundwater levels, soil moisture levels, vegetation densities, and other factors controlling surface runoff potential. March 1990 had the highest load and the highest surface runoff in the 1990-1996 period (0.5 inches). Load, runoff, and precipitation were also relatively high in June 1990. During responsive months (January-July), the base-flow load of 100-200 kg/month increased to >400 kg/month when precipitation and/or surface runoff were relatively high.

It is possible that higher concentrations and loads measured at watershed sites in 1990 partially reflected drought conditions in the previous year. Annual rainfall in 1989 was 25.6 inches, compared with 35.8 inches in 1990. Nutrients accumulating on land surfaces and/or in stream channels during dry years are scoured and transported downstream in subsequent wet years. Similar patterns have been observed in St. Paul water supply watersheds (Walker et al., 1989; Walker, 1992).

Estimates of yearly flows and phosphorus loads for 1975-1996 are listed in Appendix B and plotted in Figure 10. To develop these estimates, the daily flow record for the Platte River between 1990 and 1996 has been correlated with the daily record for the Jordan River (inflow to Lake Charlevoix), which has a continuous record for the 1975-1996 period and similar hydrology (large baseflow component). As shown in Appendix B, a drift in the correlation between Platte and Jordan River flows occurred between mid 1994 and early 1996, when Platte River flows were consistently below predicted values. Further analysis indicates that this departure cannot be explained based upon variations in precipitation. The estimated Platte River flows for 1975-1989 have been used to estimate subwatershed flows and loads using the FLUX algorithms described above. Limited historical stream concentration data (from 1980, 1987, and 1988) have been used to supplement the 1990-1996 data in calculating historical loads. Results provide approximate perspectives on flows and loads experienced in 1990-1996 relative to a longer climatologic record and relative to historical hatchery loads. Because of the observed drift in the flow model in 1995, the model may over-predict basin flows and loads in relatively low-flow years.

The average watershed flow and load in 1975-1996 (164×10^6 m^3/yr, 2232 kg/yr) were similar to average values in 1990-1996 (166×10^6 m3/yr, 2239 kg/yr). The highest watershed load occurred in 1990 (2892 kg/yr vs. 1613-2718 kg/yr for remaining years). Between 1990 and 1996, the total load decreased from 2914 kg/yr to 2022 kg/yr as a consequence of decreases in hatchery and watershed loads. This period of intensive lake and watershed monitoring was fairly representative of the range of flows and loads estimated for the entire 22-year period.

Watershed flows and loads are most responsive to precipitation in June, when water tables and soil moisture levels are relatively high. Figure 11 plots watershed flows, loads, and flow-weighted-mean concentrations against precipitation for this month. Different symbols are used to distinguish years with and without monitored stream concentrations. Flow is strongly correlated with precipitation over the entire range. Load and concentration increase substantially when monthly precipitation exceeds ~4 inches. The observed precipitation in June 1990 (6.5 inches) was exceeded in only 2 other Junes between 1895 and 1996. The extreme load measured in this month (427 kg) apparently has a low frequency of occurrence.

The 1975-1996 watershed load time series is used below in simulating lake responses to historical and potential future hatchery loading regimes. Because of apparent decreasing concentration trends detected in the North Branch and Brundage Creek, the historical load time series may over-estimate future watershed loads. To account for these trends, a second load time series has been generated with loads from the North Branch and Brundage Creek adjusted to current (1996) conditions. Adjustments are based upon trend magnitudes estimated by FLUX diagnostic procedures. Historical and de-trended monthly load time series are listed in Appendix G. The de-trending procedure reduces the average annual load from the watershed from 2232 to 2053 kg/yr. Both the historical and de-trended series are used below in simulating alternative future hatchery load scenarios (see Mass-Balance Modeling).

Lake Data Analysis

The reduction and analysis of Platte Lake water quality data described in this section are supported by the following Appendices:

- **Appendix C - Reduction of Lake Water Quality Data**. Calculation of daily, monthly, & yearly volume-weighted-mean phosphorus concentrations; two-way analyses of variance (year x month) for total phosphorus, chlorophyll-a, and Secchi depth.

- **Appendix D - Epilimnetic & Hypolimnetic Time Series**. Volume-weighted-mean epilimnetic (0-30 ft) and hypolimnetic (60-90 ft) values in 1990-1996 are plotted against sampling date. The vertical averaging scheme is designed to generally reflect surface and bottom water quality; actual thermocline depths vary seasonally and yearly.

- **Appendix E - Contour Diagrams**. Concentration isopleths (lines of constant concentration) are plotted against depth and season for each year between 1990 and 1996. Results from the 1982 study (MDNR, 1983) are also shown. Secchi depths on each sampling date are indicated by vertical bars at the top of each diagram. Line thickness increases with concentration.

- **Appendix F - Trends in Lake Water Quality**. Monthly or biweekly median concentrations for various parameters, years, depths, and seasons are plotted against time. Trend statistics are shown on each plot and tabulated separately

Additional displays of lake data (Figures 12-33) are discussed in the following sections.

Lake Data Reduction

The Court Master (CM) uses the annual, volume-weighted-mean phosphorus concentration to track lake conditions relative to the 8-ppb goal established by the Court (Brown, 1988ab). In the source data file (LAKE.XLS), the following algorithm is used to calculate a volume-weighted mean phosphorus for each sampling date and year:

1. Screen the data for "outliers" using the 15% criterion discussed above (see Data Sources); ~2.5% of samples deleted; higher values selectively deleted; 85% of deleted samples exceeded 8 ppb vs. 56% for remaining samples.

2. Calculate the mean concentration at each depth (average of <=3 replicates).

3. Calculate the volume-weighted-mean concentration on each date using a weighted mean of the average values at each depth. Prior to March 1993, concentrations in the top volume interval (depth interval = 0 to 7.5 feet) were represented by the 0-foot samples. In and after March 1993 (when sampling at 7.5 feet was initiated), concentrations in the top volume interval were represented by

the 7.5-foot sample. Surface samples, though collected, are apparently not used in calculating the volume-weighted-means after March 1993.

4. Calculate annual mean by interpolating volume-weighted-means between adjacent sampling dates and averaging over all days in year.

One concern with the procedure is that the 15% screening criterion selectively deletes high values and has apparently not been justified in the context of the sampling and analytical procedures used in this study. Generally, deletion of sample values without an objective basis (e.g., statistical outlier detection, independent evidence of contamination or laboratory error) is not recommended, particularly when the action could influence a determination of compliance. Shifting from the 0 to 7.5 foot sample to represent the top volume interval midway through the data set may introduce a bias in comparing volume-weighted-means between the two periods.

A modified data-reduction algorithm has been designed to accomplish the following objectives:

1. to generate robust and unbiased results without deleting specific data points;

2. to utilize all of the valid measurements; and

3. to provide estimates of uncertainty in the annual and long-term mean values.

Portions of the procedure have been taken from PROFILE, a computer program developed for the U.S. Army Corps of Engineers for use in reducing reservoir monitoring data (Walker, 1996). Results at daily, monthly, and yearly intervals are summarized in Appendix C. With the exception of the volume-weighting step, the same procedure is applied to Secchi depth and chlorophyll-a measurements. The procedure is described below:

1. Delete samples with supporting evidence of contamination. Field notes contained in the LAKE.XLS file attributed elevated concentrations in surface samples during February-March 1996 to decomposition products from dead fish trapped in surface ice. Substitute sample concentrations from the 7.5-foot depth on these dates.

2. Take the <u>median</u> of 3 replicates from each depth and date. This step provides a filter for anamolous values. By using the median in place of the mean, one sample out of three replicates can be contaminated without having a large effect on the results.

3. Subject the median value from each depth interval and date to an outlier test for deviation from a lognormal distribution. The Maximum Normalized Residual test (Snedecor & Cochran, 1989) yields a rejection limit of 44 ppb for 0-7.5 foot

samples at the .05 significance level. A single median value is rejected (3/19/91 surface sample, range of 3 replicates = 259-305 ppb). Substitute the 15-foot value on the same date for the rejected value.

4. Estimate values for two missing samples by interpolating between adjacent sampling dates (June 9, 1992, 75 & 90-ft samples).

5. Sampling at the 7.5-foot level was initiated in March 1993. Estimate missing values for this depth prior to March 1993 by interpolating between the 0- and 15-foot depths.

6. Calculate the volume-weighted-mean concentration on each date using a weighted mean of the median values at each depth (C-2 to C-5). Weights are derived from the lake hypsiographic profile (cumulative volume vs. depth).

7. Calculate the volume-weighted-mean concentration for each month by averaging the volume-weighted-means across sampling dates (C-6 to C-7).

8. There is one missing month (February 1990) in the 7-year data set. Estimate a mean concentration for this month by interpolating between adjacent months.

9. Calculate annual mean concentrations by averaging the monthly means within each year (C-8).

10. Conduct a two-way analysis-of-variance (ANOVA) with monthly & yearly effects. Calculate the standard error of the annual mean from the variance in the monthly ANOVA residuals and the number of sampled months (C-9).

Steps 7-9 are used in place of the CM's interpolation algorithm to provide an estimate of the annual mean that is reasonably independent of variations in sampling frequency across months. The monthly blocking procedure also facilitates estimation of variance components and uncertainty in the annual means.

Page C-8 lists yearly-mean phosphorus concentrations (for each depth interval and volume-weighted), transparency, and chlorophyll-a concentrations. The modified data-reduction procedure yields estimates of yearly-mean phosphorus concentration which are similar to those obtained using the CM's procedure:

Year	90	91	92	93	94	95	96	90-92	93-96	All
CM	9.06	7.87	8.31	7.76	7.88	8.17	7.22	8.41	7.76	8.04
Revised	9.28	7.83	8.46	7.82	7.67	8.24	7.25	8.52	7.75	8.08
Std Error	0.34	0.29	0.24	0.24	0.28	0.33	0.16	0.16	0.13	0.10

By removing fixed seasonal effects, the ANOVA model reduces the standard errors of the annual means. relative to values calculated directly from the monthly means (C-9).
Given the scale of the standard errors, the calculated annual mean values can be rounded to the nearest 0.1 ppb without loosing precision.

Comparisons of Lake Phosphorus Concentrations with 8 ppb Goal

A one-tailed t-test has been applied to compare annual-mean phosphorus concentrations with the 8-ppb goal (Table 1). Means and standard errors have also been computed for the entire period and for two year intervals (1990-1992, 1993-1996) corresponding to different hatchery loading regimes. A null hypothesis that the annual mean is less than 8 ppb is rejected in two years (1990 and 1992) and for the 1990-1992 interval. A null hypothesis that the annual mean is greater than 8 ppb is rejected in one year (1996) and for the 1993-1996 interval. For the remaining years (1991, and 1993-1995), results are inconclusive.

It can be concluded that the measured mean values were not significantly different from 8 ppb in 1991, 1993, 1994, and 1995, significantly above 8 ppb in 1990 and 1992 and significantly below 8 ppb in 1996. The apparent reduction in volume-weighted-mean between 1990-1992 and 1993-1996 can be partially attributed to reductions in hatchery phosphorus load, although reductions in watershed loads also occurred between these two periods (see Watershed Data Analysis).

Using the pooled estimate of standard error (0.29 ppb), the 90% confidence interval for the annual mean is typically ± 0.5 ppb of the measured mean. In order to state with 95% confidence that the yearly mean is below 8 ppb, the measured mean would have to be below 7.5 ppb. Similarly, in order to state with 95% confidence that the yearly mean is greater than 8 ppb, the measured mean would have to be greater than 8.5 ppb. When the measured mean is between 7.5 and 8.5 ppb, the true mean is not significantly different from 8.0 ppb.

Uncertainty in the long-term average (LTA) phosphorus concentrations estimated based upon data from 1990-1996 is considerably greater than uncertainty in the means estimated for specific years or year intervals. For example, the mean value of 8.08 ppb for 1990-1996 has a standard error of 0.10 ppb if it is used to estimate the average concentration which occurred within these years and a standard error of 0.25 ppb if it is used to estimate the LTA concentration. The LTA concentration reflects the true underlying frequency distribution of annual values. Direct measurement of the LTA is not possible, since it would require an infinite number sampling years under similar loading or watershed-management scenarios. The LTA and its uncertainty can be estimated, however, from year-to-year variations in the annual mean. Uncertainty in LTA concentration is controlled by year-to-year variations in climate and other random factors influencing lake response. As demonstrated in Table 1, null hypotheses comparing the 8 ppb goal with LTA phosphorus concentrations estimated for 1990-1996, 1990-1992, and 1993-1996 conditions are inclusive. Generally, longer data sets are required to obtain precise estimates of LTA concentration under a given loading or management scenario.

May-September means are more relevant than annual means as indicators of trophic state. Lakes are typically classified and goals established based upon growing-season measurements (Heiskary & Walker, 1988; Carlson, 1977). Phosphorus variations in late fall or winter months would be expected to have little impact on algal growth, water transparency, or recreational uses. Table 2 compares measured May-September mean concentrations with the 8-ppb goal. In four cases, the measured means were significantly below 8 ppb (1993, 1996, 1990-1996, and 1993-1996). In no cases were the measured means significantly above 8 ppb.

Given the scale of annual and seasonal phosphorus variations in Platte Lake, detection of changes and comparisons of with the 8-ppb goal require extraordinarily accurate and precise data. Few laboratories can actually provide the 0.1-ppb resolution represented in the data set. Rigid QA/QC and consistent data-reduction procedures are needed to ensure that apparent year-to-year variations do not result from drift (inaccuracy) in sampling, analytical, or calculation procedures.

Analyses of Variance

ANOVA results for phosphorus, chlorophyll-a, and transparency data from 1990-1996 are listed in Appendix C. Lay transparency data from 1977-1966 are also analyzed. Estimates of year-to-year, seasonal, and random variance components are useful for the following purposes:

1. Characterizing seasonal and year-to-year variability;

2. Estimating confidence limits for annual summary and long-term summary statistics;

3. Testing for trends;

4. Designing future monitoring programs.

The two-way ANOVA model is expressed by the following equation:

Monthly Mean = Grand Mean + Yearly Effect + Monthly Effect + Residual

Variance components for 1990-1996 data summarized below, expressed as standard deviations:

		Standard Deviation			
Variable	Mean	Yearly	Monthly	Residual	Total

Phosphorus (ppb)	8.08	0.58	0.88	1.01	1.43
Chlorophyll-a (ppb)	3.06	0.53	0.78	1.02	1.37
Secchi (ft)	10.6	0.00*	1.79	2.59	3.10

Based upon Tukey's test (Snedecor & Cochran, 1989), year-month interaction terms are insignificant ($p>.05$) for each variable. Yearly and monthly terms are significant ($p < .05$) in all cases except for the yearly term in Secchi depth (*).

Monthly and yearly means are displayed in Figure 12 (1990-1996 data) and Figure 13 (1977-1996 lay transparency data). All variables exhibit strong seasonality. Average phosphorus concentrations range from ~10 ppb in November to ~6 ppb in August-September. Average transparency ranges from ~ 14 ft in February-April to ~ 7 feet in June. Seasonal variations in transparency are unusually strong. As described below, the observed rapid decline in transparency in Spring is related to precipitation of calcium carbonate or "whiting events". Monthly chlorophyll-a concentrations range from ~ 2 ppb in February-March to ~5 ppb in November. A pronounced upward shift in the chlorophyll-a values occurred when the analytical procedure was changed (from spectrophotometric to fluorometric) in 1993. A portion of the apparent year-to-year variance in chlorophyll-a may be attributed to this change in methodology.

Between 1990 and 1996, significant year-to-year variations are detected in phosphorus, but not in transparency. This indicates that year-to-year variations in average transparency, if present, are too small for detection in the presence of strong seasonal and random variance. Significant year-to-year variations are indicated in May-September mean transparencies derived from both the CM and lay monitoring programs; seasonal-average transparencies were significantly lower in 1990, 1992, 1994, and 1995, as compared with the remaining years. These variations are examined further below (see Lake Transparency Dynamics).

Trends in Lake Water Quality

Tests for long-term trends in Platte Lake water quality have been conducted using two versions of the Seasonal Kendall Test (Hirsch et al., 1982; Hirsch & Slack, 1984) using software developed in a previous project (Walker, 1991). Results are summarized in Table 3. Detailed statistics and time series plots are shown in Appendix F. Plots show data values (seasonal medians in each year) in relation to the median for the entire period (horizontal line) and apparent trend (dashed line). Three time intervals have been examined:

1976-1996	Entire Record
1982-1996	Partial Record
1990-1996	Recent Intensive Monitoring

Figure 10 shows variations in annual phosphorus loads from the hatchery and Platte River watershed between 1976 and 1996. The 1982-1996 interval has been examined because it reflects the period after the first major reduction in hatchery load (from >1100 kg/yr in 1976-1981 to <580 kg/yr in 1982-1996. Loads were further reduced to <130 kg/yr after 1992. Loads from the Platte River watershed (exclusive of hatchery) also decreased between 1990 and 1996, primarily as a consequence of climatologic variations.

The long-term data set originates from several sources and is relatively sparse prior to 1990. Because of data sparseness and potential inconsistencies in sampling and analytical procedures, conclusions based on pre-1990 data are qualified. Generally, at least ten years of data are desired for detecting trends in water quality data (Hirsch et al., 1982). For shorter records (e.g., 1990-1996), there is greater risk that an apparent trend is attributed to a pattern which appears to be a trend, but is actually a random occurrence (Type I error) and there is greater risk that actual trends will go undetected, particularly if they are small (Type II error). Although the 1990-1996 data set appears to be ideal with respect to sampling intensity and uniformity, its duration is less than ideal for detecting trends. Power for detecting long-term trends will increase substantially with a few more years of intensive monitoring, provided that consistent sampling and analytical procedures are maintained.

Tested variables include total phosphorus, chlorophyll-a, transparency, dissolved oxygen (D.O.), temperature, and dissolved oxygen deficit. The last variable (saturation D.O. - observed D.O.) adjusts measured D.O. values for variations in temperature. Phosphorus concentrations at each sampled depth have been tested, along with the epilimnetic average (volume-weighted-mean 0-30 feet), hypolimnetic average (volume-weighted-mean, 60-90 feet), and volume-weighted-mean for the whole water column (using both data-reduction procedures methods described above).

The test accounts for seasonality by dividing each year into a number of equal-length intervals or seasons. Median values are calculated for each season and year. A null hypothesis of no trend is tested by making year-to-year comparisons of measurements within each season. The appropriate season length depends upon sampling intensity. A monthly interval (12 seasons per year) is used for testing trends in annual values. For 1990-1996, separate series of tests have been conducted for trends within certain portions of the year (May-September, October-March, April-June, and July-September). To reflect nominal sampling frequencies, these tests are based upon monthly intervals for October-March and biweekly intervals for the remaining periods. Tests for trends in Lay monitoring data (Secchi depth) are based upon weekly intervals between May and September.

Appendix F shows linear trend lines in relation to the data. The estimated trend slope equals the median time derivative based upon all year-to-year comparisons within each season. Probabilities estimated by the test assume a monotonic (generally increasing or generally decreasing), but not necessarily linear trend.

Advantages of the Seasonal Kendall test relative to other trend-detection procedures are that it is (1) non-parametric (i.e. does not require normally distributed data), (2) robust to the presence of outliers, and (3) does not require complete data sets (a reasonable number of missing values are permitted). The second version of the test (Hirsch & Slack, 1984) differs from the first (Hirsch et al, 1982) in that it accounts for serial correlation in the observations. The second test reduces the chance of finding "false trends" when the data are serially correlated. It is less powerful than the first test (i.e., may fail to detect real trends), when serial correlation is absent, however. In this sense, the second test is a more conservative one. For this reason, results from both versions of the test are reported, along with a measure of serial correlation (first-order serial correlation of de-trended seasonal medians).

Table 3 lists trend magnitudes for each data set in which the null hypothesis is rejected at $p<.10$ based upon the first version or second version of the test. Trend magnitude is expressed as a percent of the median value per year. Single and double asterisks (*,**) identify cases where the second (more conservative) version of the test yields p values <.10 and <.05, respectively. Detailed test results are listed in Appendix F. Results are grouped in two series of tests: long-term (1976-1996, 1982-1996) and recent (1990-1996) time intervals.

Results of long-term trend analyses are generally consistent with declining trends in phosphorus loads from the hatchery (Figure 10). Decreasing trends in volume-weighted-mean phosphorus concentrations are indicated for each time period (1976-1996, 1982-196, 1990-1996). Trend slopes range from -1.4 to -2.9 %/year. Time series plots indicate relatively high variance in the phosphorus data prior 1990. This partially reflects relatively low sampling intensities and possible variations in sampling or analytical procedures. Decreasing phosphorus trends are accompanied by increasing trends in annual-mean transparency (1.5 to 1.8 %/yr). The 1990-1996 transparency trend is not confirmed by the second version of the test, however. Increasing trends in May-September Secchi depths from the lay monitoring program are indicated for 1976-1996 (3.6 %/yr) and 1983-1996 (2.8%/yr), but not for the 1990-1996. No long-term trends are indicated for chlorophyll-a or dissolved oxygen at 90 feet.

More detailed analyses of recent data reveal differences in trend magnitudes as a function of depth interval and season. The following seasons are examined separately:

All	All Months
May-September	Growing Season
October-April	Non-Growing Season
April-June	Spring
July-September	Summer

Generally, phosphorus trends are stronger in the winter (October-April) and spring (April-June) intervals than during the growing season/summer and stronger at the bottom of the lake than at the top. Phosphorus trends in May-September or July-September are indicated only at 90 feet depth. Trends in surface phosphorus concentration (0 feet) are not detected in any case.

When the data set is restricted to 1991-1996, phosphorus trends are indicated for volume-weighted-mean and 45-foot value, but neither of these is confirmed by the second version of the test. It is possible that trends detected for the 1990-1996 period primarily reflected reductions which occurred between 1990 and 1991 and largely resulted from unusually high runoff P loads in 1990 (Figure 10). Shortening the time interval weakens the tests, however (i.e. makes it more difficult to detect trends of these magnitudes).

Recent trends in transparency are restricted to the non-growing season (October-April, +2.5%/yr). Trends during the growing/recreational season (May-September) are detected in neither the CM nor the lay transparency data from 1990-1996. As shown in a subsequent section, the high year-to-year variance in May-September transparency values is controlled by whiting events driven by variations in temperature, vertical stratification, and phosphorus. Improvements in transparency during the recreational season resulting from reductions in phosphorus load over the 1990-1996 period (Figure 10) are not detectable in the presence of climatologic and other random variations.

The apparent increasing trend in May-September chlorophyll-a concentrations (7.3 %/yr) is not confirmed by the second version of the test. It is possible that the apparent trend is an artifact caused by the change in analytical procedure during 1993 (see Data Sources).

Improving water quality in the hypolimnion is indicated during the critical April-June period, when vertical stratification develops and oxygen concentrations begin to decrease. Trends in hypolimnetic phosphorus (-3.1%/yr), dissolved oxygen (+2.3 %/yr), and dissolved oxygen deficit (-9.3 %/yr) are all indicative of improving conditions. A decreasing trend in hypolimnetic temperature (-2.3 %/yr) is also suggested, but not confirmed by the second version of the test. It is possible that improvements in phosphorus and dissolved oxygen are related to temperature; cooler temperatures would slow the rate of oxygen depletion. The temperature trend is determined largely by the relatively high temperatures reported in 1990 (both in the epilimnion and hypolimnion). When data from 1990 are excluded from the April-June trend analyses, a temperature trend is not indicated, but trends in hypolimnetic phosphorus (-4.6%/yr), dissolved oxygen (+2.8 %/yr), and dissolved oxygen deficit
(-12.6%/yr) remain. These apparent improvements are not explained by variations in temperature and may reflect a declining trend in spring algal productivity over the 1990-1996 period. This may indicate that productivity was responding to reductions

in phosphorus loads from the hatchery and watershed that occurred between 1990 and 1996 (Figure 10).

Lake Transparency Dynamics

Controlling Factors

Lake transparency is typically lowest during algal pulses and whiting events in the late Spring and early Summer (Figure 12). Based upon the Court Master's data (1990-1996), the average Secchi depths in June is ~7 feet, as compared with 9-14 feet in other months. The same seasonal pattern is evident in the Lay monitoring data (Figure 13).

The seasonal minimum in transparency is controlled by calcium carbonate precipitation. Figure 14 shows isopleths of pH, temperature, and calcium carbonate saturation index (SI) for March 1996 - February 1997. SI values have been obtained from the Court Master's LAKE.XLS file. SI values above zero indicate supersaturated conditions with respect to calcium carbonate, which are conducive to calicite precipitation. SI values below zero are conducive to dissolution of calcite and release of adsorbed phosphorus. Indices below zero are found in the bottom of the lake between June and September 1996 and throughout the water column in February 1997. Sensitivity analysis of the equations used to calculate the SI indicate that the observed values are driven more by variations in pH than by variations in water temperature.

In typical hardwater lakes, algal cells associated with spring blooms and calcium carbonate precipitation would be expected to control the seasonal minimum in transparency (Wetzel, 1996). Spring algal blooms reflect seasonal increases in water temperature and solar radiation, formation of the thermocline, and elevated nutrient concentrations associated with lake turnover and phosphorus loading from spring runoff. Calcium carbonate precipitation is triggered when solubility decreases as a consequence of increases in temperature and/or pH. By removing carbon dioxide from the water column, spring algal pulses can cause increases in pH and thereby foster calcium carbonate precipitation. Climatologic variations drive the entire process by influencing water temperature, thermocline depth, and photosynthesis rates. Nutrient limitation of algal populations and photosynthesis rates represent linkages to nutrient concentrations and loads (Schelske & Hodell, 1991; Shelske, 1991).

The inverse of the Secchi depth is proportional to the light extinction coefficient, which measures absorption and scattering of light by dissolved and particulate materials in the water column (Walker, 1985a). Based upon the relatively low seasonal range of chlorophyll-a values (2 - 5 ppb), algal biomass typically accounts for a small portion of light extinction in Platte Lake. Figure 15 plots monthly-mean light extinction coefficients against chlorophyll-a concentration, based upon data from 1994-1996 when chlorophyll-a concentrations were determined fluorometrically. The following equations partition total light extinction into three components:

$$E = \text{Light Extinction Coefficient } (m^{-1}) = 1.66 \,/\, \text{Secchi Depth (m)}$$

$$E = \text{Background} + \text{Algal Component} + \text{Residual}$$

$$E = 0.27 + 0.042 \text{ Chl-a (ppb)} + \text{Residual}$$

The algal component can be modeled as a linear function of chlorophyll-a concentration with a proportionality constant of ~0.042 m^{-1} ppb^{-1} (Walker, 1985a, 1996).

The background component (0.27 m^{-1}) accounts for light extinction by dissolved color and background particulate materials. It is calibrated using data from March, the month with the lowest non-algal light extinction. The background estimate corresponds to a maximum transparency of ~20 feet in the absence of algae or residual light extinction. The residual term is estimated by difference from the other terms using observed Secchi depth and chlorophyll-a values. This term measures seasonal variations in calcium carbonate particles and other non-algal dissolved and particulate materials. A portion of the background light extinction may also be attributed to calcite particles (to the extent that they are present in March).

Based upon this partitioning scheme, the percentage of total light extinction attributed to algae varies from a maximum of 39% in December to a minimum of 15% in June. The June minimum corresponds to whiting events, which are reflected by dramatically higher total light extinction (0.98 vs. 0.34-0.65 m^{-1}) and residual light extinction (0.56 vs. 0.0-0.25 m^{-1}). It is likely that the seasonal minimum in transparency is attributed primarily to light scattering by inorganic calcite particles, rather than absorption and scattering by algal cells. As noted by Wetzel (1996), whitings are often mistakenly perceived by the public as algal blooms. Although algal photosynthesis may play a role in these events (by increasing pH), the algal cells themselves account for a relatively small proportion of the total light extinction during these events.

Figure 16 plots total and algal light extinction as a function of time, using data from sampling dates with measured values for chlorophyll-a and Secchi depth. Relatively intense whiting events were experienced in 1990, 1992, 1994, and 1995. During the June 1995 event, a minimum Secchi depth of 2 feet was recorded. In each event, algae accounted for a small percentage of the total light extinction. It is possible, however, that algal photosynthesis played an important role in these events by raising pH and promoting calcite precipitation.

Figure 17 shows monthly-average values for transparency and related lake, watershed, and climatologic variables in each year. These displays elucidate differences between years with relatively intense whiting events (solid lines) and years with relatively mild events (dashed lines).

The role of algae is indicated by the fact that the occurrence of intense whiting events is associated with higher concentrations of chlorophyll-a (using either analytical procedure) and phosphorus. Decreases in alkalinity (0-15 ft averages) between May and July (one measure of algal productivity) were more pronounced and pH levels were higher in 1994 and 1995, as compared with 1993 and 1996 (no data available for 1990-1992).

The roles of temperature and thermal stratification are more difficult to discern directly from Figure 17. As demonstrated below, however, May-June Secchi depths are negatively correlated with mixed-layer water temperatures, when controlled for variations in phosphorus concentration. The extreme event in 1995 was associated with relatively strong thermal stratification and high water temperatures that, in turn, reflected relatively low wind speeds, high air temperatures, and clear weather in late May and early June. This event is discussed in more detail below.

Direct correlations between monthly transparencies and monthly phosphorus loads from the watershed or hatchery are not evident in Figure 17. Over longer time scales, however, phosphorus loads are likely to be important because they influence lake P concentrations and algal productivity. A mass-balance model is developed below to express these relationships.

<u>Regression Model for May-June Transparency</u>

The regression model in Figure 18 indicates that ~60% of the variance in May-June Secchi depths can be explained based upon variations in mixed-layer phosphorus concentration and water temperature. The model is of the following form:

$$1/S = 0.06 + 0.000828 \; (P)^{1.047 + 0.0618\,T}$$

where,

S = Secchi depth (feet)
P = Mixed Layer Total P Concentration (ppb)
T = Mixed Layer Water Temperature (deg-C)

Observed and predicted values are plotted against sampling date in Figure 19. The model has been calibrated to data from 38 sampling dates between May 3 and July 3 of 1990-1996. Standard error (~29% of predicted value) increases substantially when the calibration period is extended beyond the above date range. The model under-estimates transparencies when applied to data from July and August, after the peak whiting period.

The ranges of phosphorus and transparency values represented in the data set are shown in Figure 20. Overall calibration ranges are 4 to 14 ppb for phosphorus and 8 to 24 deg-C for temperature. The phosphorus range of 7-9 ppb and temperature range of 8-22 deg-C is well-represented in the data set. Observation frequencies and model reliability are lower outside of these intervals.

Predicted responses to phosphorus and temperature are shown in Figures 21 and 22. Partial residuals plots (Figure 21) show the effect of phosphorus at a constant temperature of 20 deg-C and the effect of temperature at a constant phosphorus concentration of 8 ppb. Figure 22 shows two-dimensional response surfaces for the mean and standard error of predicted transparency.

The intercept of the equation (0.06 ft^{-1}) accounts for background turbidity and color; it corresponds to a maximum transparency of 16.7 feet in the absence of phosphorus. The model indicates that transparency sensitivity to phosphorus concentration increases with water temperature. This sensitivity is reflected by the fact that the phosphorus exponent varies from 1.54 to 2.53 over a seasonal temperature range of 8 to 24 deg-C. This increasing sensitivity may reflect the controlling effects of phosphorus on algal densities and the dependence of photosynthesis (CO_2 production) rates on algal densities and water temperature.

The model consistently over-predicts measured transparencies in 1995. As indicated in Figure 23 and Appendix E, the vertical temperature structure was unusual in June 1995. There was a pronounced vertical temperature gradient near the surface. On the date of minimum transparency (June 20, 1995, 2 feet), there was a 10-degree difference between the water surface temperatures at 0 and 15 feet (24 vs. 14 deg-C). The average temperature gradient between 0 and 30 feet (0.7 deg-C/ft) was much higher than that experienced on other sampling dates in May-June (range 0 to 0.3 deg-C/ft). A shallower thermocline and warmer surface temperature would be conducive to photosynthesis. For a given solar condition, the average light intensity in the mixed layer increases as the thermocline depth decreases. Photosynthesis and CO_2 production rates would be expected to increase with water temperature and light intensity. The shallow thermocline in June 1995 developed during of a period of warm, sunny, and calm weather in late May and early June. As shown in Figure 17, June 1995 had higher average temperature, more hours of sunshine and lower wind speeds, as compared with Junes of other years.

A shallow thermocline also developed in May-June 1991 (Appendix E, Figure 23), but was less pronounced than that experienced in May-June 1995 (yearly minimum transparency 7 feet on May 3, 1991 vs. 2 feet on June 20, 1995). Another important distinction is that the thermocline developed approximately 3 weeks earlier in 1991 (May 3-14, 1991 vs. May 23-June 6, 1995). With the onset of stratification, mixed-layer phosphorus concentration decreased from 8.2 ppb at spring overturn (April 16, 1991) to 4.5 ppb on May 28 as a result of algal uptake and sedimentation. In 1995, mixed-layer P concentrations decreased from 9.1 ppb at spring overturn (April 11) to

8.4 ppb on May 23. As a consequence of weaker stratification in May 1995, there was more phosphorus in the mixed layer at the beginning of June, when the surface waters warmed up rapidly under sunny, calm weather and the intense whiting event began. Climate and physical factors largely controlled the differences in lake response during these two years. These overwhelmed any influences of phosphorus loads from the hatchery (39 kg/month in May-June 1991 vs. 9 kg/month in May-June 1995) or from the Platte River watershed (260 vs. 160 kg/month, including hatchery).

Transparency vs. Phosphorus over Various Time Scales

This section explores relationships between May-September transparency and volume-weighted-mean phosphorus concentration. From a water-quality management perspective, lake transparencies between May and September are most relevant for protecting aesthetic qualities and recreational uses. The volume-weighted-mean phosphorus concentration has been adopted for tracking lake conditions relative to the 8 ppb goal (Fuss, 1989). The volume-weighted-mean concentration is also convenient from a modeling perspective because it can be linked to watershed loads using relatively simple mass-balance models.

Relationships between May-September light extinction coefficients and volume-weighted-mean phosphorus concentrations averaged over daily, monthly, seasonal (May-September), and yearly time scales are shown in Figure 24. Corresponding relationships between Secchi depth and phosphorus are shown in Figure 25. Responses to phosphorus are evident at each time scale. On monthly time scale, the light extinction coefficient typically varies between 0.4 and 0.6 m^{-1} when P concentrations are less than 8 ppb. The extinction coefficient generally increases to values exceeding 1 m^{-1} at P concentrations exceeding 10 ppb. The response above 8 ppb is highly variable, apparently because of the controlling effects of water temperature and vertical stratification on algal growth and the calcite precipitation process. The highest monthly-average extinction coefficient (1.7 m^{-1}) occurred in June 1995, when the average total P concentration was 8.2 ppb. As discussed above, high water temperature and a shallow thermocline amplified this response.

Variance in the response generally decreases as the averaging time scale increases from daily to seasonal. The May-September volume-weighted-mean P concentration is a good predictor of both the seasonal-average ($r = -0.72$) and the monthly minimum ($r = -0.89$) Secchi depth. Years with relatively mild whiting events (1991, 1993, 1996) had minimum monthly-average transparencies between 8 and 10 feet and May-September average P concentrations between 6.9 and 7.3 ppb. Years with relatively intense whiting events (1990, 1992, 1994, and 1995) had minimum monthly-average transparencies between 3 and 5 feet and May-September average P concentrations between 7.7 and 8.4 ppb.

With a threshold P concentration of 8 ppb, the annual, volume weighted-mean P concentration correctly predicts three out of the four years with relatively intense whiting events. The exception is 1994, when the yearly volume-weighted-mean was 7.7 ppb and the minimum transparency was 5 feet. Average and minimum transparencies are more strongly correlated with the May-September average P concentration (r = -0.72 and -0.89. respectively) than with the yearly average P concentration (r = -0.43 and -0.64, respectively).

Similar patterns are evident in historical data from Platte Lake (1976-1989) and Lay transparency monitoring data from 1990-1996. Figure 26 plots extinction coefficients against P concentration for each time scale. Figure 27 plots Secchi depths against P concentration for each time scale.

Historical phosphorus data are limited by the number of samples, lower analytical precision, and lower sampling intensity. Sufficient historical data are not available for evaluating monthly, seasonal, or yearly time scales. On a daily basis, historical data suggest a threshold effect involving increases in both the mean and the variance of light extinction coefficient at phosphorus concentrations exceeding 7-8 ppb (Figure 26). The effect is stronger in May-June samples than in July-September samples. These patterns are generally consistent with patterns in 1990-1996 data.

Monthly-average transparencies from the Lay monitoring program in 1990-1996 have been paired with monthly-average transparency and phosphorus data CM's monitoring program. Figure 28 shows that Lay and CM transparency data are highly correlated, but not equivalent. Differences in methodology may account for the fact that the Lay measurements are typically 1-4 feet higher than the CM's measurements. Generally, correlations with phosphorus based upon the Lay transparency data (Figures 26-27) are similar to those based upon the CM's data (Figures 24-25). The May-September average P concentration is superior to the yearly P concentration as a predictor of seasonal mean and minimum transparencies, based upon both sets of transparency measurements.

Long-Term Variations in Transparency & P Load

Figure 29 plots long-term variations in Secchi depth measured under the lay monitoring program in relation to hatchery P loads between 1977 and 1996. In other regions, transparency has been directly correlated with user-perceptions of aesthetic qualities and suitability for recreational uses (Heiskary & Walker, 1988, Smeltzer & Heiskary, 1990). Direct measurements of phosphorus load from the hatchery are available for 1981-1996. Estimates of pre-1981 loads are based upon reports by Grant (1979), Kenaga & Evans (1982), and scaled against annual fish production data provided by MDNR.

Increases in transparency over the 1975-1996 time frame are generally correlated with decreases in hatchery P load. Apart from random variations attributed to climatologic factors, long-term changes in non-hatchery phosphorus loads may also have occurred within the same time frame. As discussed above, FLUX calculations indicate reductions in loads from the North Branch of the Platte River and Brundage Creek within the 1990-1996 time frame (adjusted for variations in flow). Increases or decreases in loads attributed to watershed development, watershed management, and/or changes in fish management (influencing potential P contributions from migrating salmon) may have also occurred but are not readily quantified.

Figure 29 suggests that there is a time lag in the transparency response to reductions in hatchery load. For example, the major reduction in load between 1981 and 1982 was followed by a gradual increase in transparency between 1982 and 1984. Similarly, the load reduction between 1988 and 1989 was followed by a transparency increase in 1990. Transparency responses to load reductions between 1990-1992 and 1993-1996 are not apparent in Figure 29 or in results of trend analyses (Appendix F). As discussed below (see Phosphorus Recycling Mechanisms), the lag in transparency response may reflect influences of phosphorus recycling from lake sediments.

Compared with yearly maximum and average transparencies, the yearly minimum transparency has been less responsive to reductions in load (Figures 13 & 29). The lowest transparency on record (2 feet) was measured in June 1995. As discussed above, the whiting event in this month was apparently amplified by a relatively high surface-water temperature, a shallow thermocline, and rapid warming in late May. The long-term transparency time series suggests that climatologic conditions conducive this extreme response occur at a low frequency.

Appendix C summarizes the transparency distribution in each year based upon the mean, standard deviation, range, and frequency of values less than 12, 10, 8, and 6 feet. Given the high seasonal variability in transparency, user perception of water quality and use impairment are more likely to be correlated with the frequency of extreme or nuisance conditions than with seasonal-average conditions (Walmsley, 1984; Walker, 1985b). Frequency statistics are plotted against year in Figure 30.

Because of the strong seasonal variance in transparency (Figure 13), means and frequency statistics may be biased in years with missing observations in one or more weeks. To reduce this bias, missing values have been estimated using a two-way analysis of variance model (Appendix C) prior to computing frequency statistics. This procedure has a small effect on the analysis because the frequency of missing values is low (7.8% overall and 1.3% during June, when seasonal minimum transparency values have generally been observed).

One of the bases for the court-imposed 8-ppb phosphorus goal was evidence of a significant decline in transparency, relative to conditions existing prior to initiation of the hatchery's salmon program in 1965. The following excerpt is from Judge Brown's Opinion (1988a):

> *"14) Prior to the establishment of the salmon program:*

a) The waters of Big Platte were clear, objects being distinguished at depths of 10-12 feet;...

15) Subsequent to the operation of the salmon program:
a) The waters of Big Platte have become mucky and obscure;..."

A data summary presented by Canale et al. (1991) indicates a Secchi disk reading of 3 meters (~10 ft) in 1940. A transparency of 10 feet is a criterion for (presumably acceptable) conditions that existed prior to the hatchery salmon operation.

Figure 31 shows Secchi interval frequencies and average hatchery loads in four time intervals that bracket major reductions in hatchery load (1977-1981, 1982-1988, 1989-1992, 1993-1996). The frequency of Secchi depths less than 10 feet decreased from 96% in 1977-1980 to 27% in 1989-1992. Between 1989-1992 and 1993-1996, hatchery load was further reduced by 70%, but transparency response was small. The frequency of Secchi depths less than 10 feet decreased from 27% to 25%. It is likely this low response reflects the fact that hatchery loads in both periods represented a relatively small percentage of the total P load to the lake (< 10%, see Lake Phosphorus Balances). Further improvements in transparency under 1993-1996 loading regimes may eventually be realized, however, because of the delay in transparency response attributed to P recycling from lake sediments. A model for simulating this response is developed below (see Mass-Balance Modeling).

Lake Phosphorus Balances

Appendix G lists water and phosphorus balances for Platte Lake formulated on monthly and yearly time scales. Results are also summarized for the entire period (1990-1996) and for periods reflecting reductions in hatchery loads (1990-1992, 1993-1996). Calculations are based upon the following fundamental mass-balance equations:

Water Balance: Outflows = Inflows + Precipitation - Evaporation

Phosphorus Balance: Outputs = Inputs - Increase-in-Storage - Net Sedimentation

Measured input terms include the Platte River at the USGS gauging station (adjusted for hatchery contributions), Dead Stream, and the hatchery discharge. Estimated input terms, developed using assumptions described by Chapra (1996), include runoff from developed areas around the lake, atmospheric inputs, shoreline septic systems, and fishponds. To reflect net contribution of urban runoff above the background watershed load, Chapra's export coefficient for local urban areas (60 kg/km^2-yr) has been reduced by the average export factor for the watershed (5.4 kg/km^2-yr), as derived from load calculations in Appendix B. While Chapra's export coefficient for urban areas seems reasonable, compilation of a current land use map for the watershed would be required to verify his estimate of contributing urban drainage area (0.91 km^2), which was based upon the number of shoreline residences.

Phosphorus output from the lake has been estimated by multiplying calculated lake outflow volumes by measured mean epilimnetic (0-30 ft) phosphorus concentrations in each month. Changes in lake surface elevation (generally ± 1 ft) have been ignored in the water-balance and mass-balance calculations. Lake water-balance terms have been estimated based upon regional monthly lake evaporation rates (van der Leeden et al., 1990) and monthly precipitation data compiled in Appendix A. The Net Sedimentation term in the phosphorus balance has been calculated by difference from the remaining terms; as such, it reflects phosphorus deposition to or from the lake sediments, as well as the cumulative effects of any measurement or estimation errors in the other mass-balance terms.

Measured loads from the Platte River above the USGS gauge, North Branch, and the hatchery accounted for 81% of the total estimated load to the lake over 1990-1996. This suggests that total load estimates are relatively insensitive to assumptions employed to estimate ungauged loads. There is greater uncertainty in the mass-balance results for June 1994 through February 1996, when stream concentrations were not measured.

The hatchery load represented 9.7% and 3.9% of the total lake load in 1990-1992 and 1993-1996, respectively. As discussed above, the standard error of 1990-1996 gauged tributary loads is on the order of 4%. Because of uncertainty in the unmonitored sources, estimates of total lake load would be expected to have higher standard errors, particularly for a given year. It is reasonable to conclude that hatchery loads in 1993-1996 were less than the uncertainty bandwidth for the total lake load.

The following table summarizes P loads and lake concentrations for 1990-1992 and 1993-1996:

	Total P Load (kg/yr)			Lake P
Period	Hatchery	Non-Hatchery	Total	(ppb)
1990-1992	316	2931	3247	8.5
1993-1996	96	2367	2463	7.7
Reduction	220	565	785	0.8
Percent Reduction	70%	19%	24%	9%

The range of non-hatchery loads (2367 to 2931 kg/yr) is comparable to the range estimated in previous studies of Platte Lake: 2358 kg/yr (MDNR, 1983), 2444 kg/yr (Canale et al., 1991), 2810 kg/yr (Chapra,1996). As a consequence of decreases in loads from the hatchery (70%) and other sources (19%), the total lake load decreased by 24% and the lake volume-weighted-mean phosphorus concentration decreased by

9% (from 8.5 to 7.7 ppb). Contrary to assumptions inherent in most empirical phosphorus-balance models, lake phosphorus concentrations did not exhibit a proportionate response to external loads. Variations in net retention efficiency are reflected by average settling rates of 20.1 and 15.1 m/yr for 1990-1992 and 1993-1996, respectively (Appendix G). Recycling of sediment phosphorus may have buffered the lake response to the reduction in external load.

The mean residence times of water and phosphorus in 1990-1996 were 5.9 and 2.8 months, respectively. The water residence time is calculated as the lake volume divided by the average outflow rate. The phosphorus residence time is calculated as the average mass of phosphorus stored in the lake water column divided by the average total load. The phosphorus residence time is lower than the hydraulic residence time because of the net flux of phosphorus from the water column to lake sediments.

Given the scale of phosphorus residence time (2.8 months), it is useful to examine lake phosphorus mass balances at a quarterly time step. Figure 32 shows phosphorus balance terms computed for 3-month rolling intervals starting January-March 1990 and ending October-December 1996. These plots reveal important seasonal aspects of lake phosphorus dynamics. Plotted terms include total input, hatchery input, total output, storage increase, and net sedimentation. A long-term declining trend in total inputs reflects reductions in both hatchery load and gauged tributary loads. Quarterly input loads vary from 400 to 1200 kg and net sedimentation rates range from -100 to 900 kg. The fact the ranges of these terms are similar (800 and 1000 kg, respectively) indicates that internal lake dynamics are at least as important as external loads in controlling seasonal variations in lake phosphorus concentration.

Figure 32 also shows quarterly variations in other factors derived from the mass balances (water & phosphorus residence times, inflow & lake phosphorus concentrations, and settling rate). Regular seasonal variations in the net settling rate are evident. Rates are typically highest during the peak of the growth season (20 to 40 m/yr in June-August) and lowest in fall (-5 to 10 m/yr in September-November). These variations reflect underlying mechanisms responsible from phosphorus removal from the water column and/or release from bottom sediments.

Maximum settling rates correspond to the period of thermal stratification, warm water temperatures, and maximum solar intensities; these conditions would promote phosphorus removal from the water column via algal uptake and calcium carbonate precipitation. As discussed above, the June minimum in transparency is correlated with phosphorus concentration and water temperature. Relatively intense whiting events in June 1992, 1994, and 1995 were associated with relatively low phosphorus settling rates. May-July settling rates ranged from 14-20 m/yr in these years compared with 28-38 m/yr in the remaining years. The higher phosphorus concentrations in those years apparently resulted primarily from variations in the net settling rate, not from variations in external loads. The event in 1990 was associated with higher external loads from the watershed and hatchery. Intense whiting events were not experienced in 1991 or 1993, despite the fact that external loads were equal to or exceeded those experienced in years with intense events.

Yearly phosphorus-balance terms derived from Appendix G are plotted in Figure 33. External loads decreased from 3606 kg/yr in 1990 to 2390 kg/yr in 1996 and lake volume-weighted-mean P concentrations decreased from 9.3 to 7.3 ppb. Annual net settling rates ranged from 12.2 m/yr in 1995 to 24.5 m/yr in 1991. Lower setting rates tended to occur in years immediately following reductions in external load. Since year-to-year variations in the settling rate are larger on a percentage basis than year-to-year variations in the external load, a model with a fixed net settling rate, similar to that developed by Chapra (1996), would not be successful in explaining year-to-year variations in lake phosphorus concentration over this period. The relatively high settling rate of 24.5 m/yr in 1991 may reflect the early warming and strong thermal stratification which occurred in that year (i.e., an extended growing season). The relatively low settling rates in 1994 and 1995 may reflect internal recycling mechanisms and/or under-estimation of external loads resulting from lack of tributary monitoring data between June 1994 and February 1996.

Phosphorus Recycling Mechanisms

The following observations suggest that P recycling from lake bottom sediments was partially controlling lake phosphorus concentrations during the 1990-1996 period:

1. P accumulation in the bottom waters of the lake during winter and summer stratified periods (Appendices D & E).

2. Similar magnitudes of external loads and net sedimentation rates quantified at 3-month intervals (Figure 32);

3. Consistently low or negative net P settling rates in the Fall (Figure 32); and

4. Lack of proportionate response of lake concentration to reductions in external load; between 1990-1992 and 1993-1996, external loads decreased by 18% and lake concentrations decreased by 9% (Figure 33, Appendix G).

Potential recycling mechanisms operating in various seasons are discussed below.

Low or negative phosphorus settling rates in fall (Figure 32) reflect a period of cooling water temperatures, lake turnover, and normally low algal activity. Chapra (1996) identified this pattern in modeling monthly variations in lake phosphorus concentration. Since the calculated net sedimentation rates are based upon volume-averaged concentrations, low net sedimentation rates cannot be explained by mixing of phosphorus from the hypolimnion into the epilimnion at fall turnover.

It is possible that lower net setting rates in fall reflect turnover of unmonitored shallower hypolimnetic basins in the Lake. The measured volume-weighted-mean P concentrations are based upon data from the western basin only (90 feet, deepest point). There is also a larger and shallower (~75 feet) basin in the center. Because it is shallower, anaerobic conditions may develop earlier and lead to greater phosphorus accumulation in this basin, as compared with the monitored western basin. The potential for this mechanism would depend upon the extent to which the two basins are hydraulically isolated. There were no significant differences in bottom phosphorus concentrations between the two basins in August 1982 (MDNR, 1983).

One plausible mechanism for fall P recycling is death and decay of rooted aquatic vegetation found in littoral zones of the lake. Phosphorus stored in plant biomass during the growing season would be released into the water column when plant biomass recedes in the fall. Lack of data on plant biomass levels and composition precludes quantitative evaluation of this mechanism.

Another plausible mechanism is wind-induced re-suspension of loose floc and/or surface sediments containing fresh calcite particles and organic debris. Figure 34 compares seasonal variations in apparent settling rate and wind speed. There is a strong seasonal maximum in wind speed and wind energy input in November (Appendix A). The cube of the mean daily wind speed (an approximate relative index of energy input) varies from ~600 knots^3 in August to ~1600 knots^3 in November. Wind effects would be amplified by the large surface area and wind fetch. Low or negative apparent settling rates in fall are followed by relatively high settling rates in the winter. High settling rates in winter (despite low algal activity and low $CaCO_3$ saturation index) may reflect settling of the resuspended particles when the water column is shielded from the wind by ice cover. Wind mixing may again contribute to lower settling rates in spring.

Canale et al. (1991) developed a model which links phosphorus recycling from the bottom sediments of Platte Lake to anaerobic conditions in the bottom waters. Under 1982 and 1990-1996 conditions, however, phosphorus accumulation in bottom waters appeared to be largely unrelated to dissolved oxygen. Phosphorus buildup generally started in January-February when bottom dissolved oxygen levels typically exceeded 8 ppm (Appendix E). The initial buildup was eliminated in April when the water column was vertically mixed. Maximum hypolimnetic phosphorus concentrations were generally observed in May and early June, well before the onset of anaerobic conditions. Buildup was generally restricted to concentrations less than 18 ppb (roughly a doubling of surface values) and to depths greater than 60 feet, which account for <10 % of the total lake volume.

Under severe anaerobic conditions, reduction of iron would be expected to trigger sediment phosphorus releases. This recycling mechanism requires negative oxidation-reduction potential (redox), a condition which is thermodynamically equivalent to dissolved oxygen values below zero. Apparently, this mechanism was relatively unimportant in Platte Lake under 1990-1996 conditions, probably because anaerobic conditions were not severe enough. Trace quantities of dissolved oxygen in the hypolimnion can be sufficient to maintain redox potential high enough to avoid iron reduction and phosphorus release. Direct measurement of iron concentrations in the hypolimnion would provide better diagnostic information. The relatively high iron/phosphorus ratio in the surface sediments of Platte Lake (~7 on mass basis, MDNR 1988 core) suggests that phosphorus and iron released into the water column as a result of iron reduction in the sediments would be tend to re-precipitate when exposed to aerobic surface waters.

The lack of correlation between phosphorus buildup and anaerobic conditions indicates that algal productivity (and phosphorus loads) during the 1990-1996 period were sufficient to avoid severe reducing conditions required to trigger phosphorus releases from bottom sediments. Increasing trends in the spring hypolimnetic dissolved oxygen concentrations in 1990-1996 (Appendix D) suggest that the potential for redox-driven recycling of phosphorus was also decreasing.

The observed phosphorus buildup at the bottom of the lake in winter and spring could be attributed to the following mechanisms:

1. Mineralization of organic phosphorus contained in particulate organic materials deposited on the lake bottom (algal cells and other detritus);

2. Dissolution of fresh calcium carbonate particles and release of adsorbed phosphorus; and

3. Diffusion from historical sediments;

The second mechanism is controlled by the solubility of calcium carbonate, which decreases with increasing temperature and pH. Phosphorus is removed from the surface waters by algal uptake and settling and by adsorption onto calcite particles that form when under conditions of high pH and temperature. The adsorption process may be partially reversed in the bottom waters under conditions of low pH and temperature. As shown in Figure 14, $CaCO_3$ Saturation Index values below zero (conducive to dissolution of calcite and release of adsorbed phosphorus) were found in the bottom of the lake between June and September 1996 and throughout the water column in February 1997.

Diffusion of phosphorus from historical sediments is a potential P recycling mechanism that could operate in any season. The MDNR has supplied unpublished data from a vertical sediment core collected in November 1988 and sectioned in 1-cm increments. A peak in sediment phosphorus content is indicated in the 2-3 cm increment (1100 mg/kg vs. 830 mg/kg at 0-1 cm and <500 mg/kg at depths > 10 cm). This suggests that there may be a positive driving force for phosphorus diffusion out of the sediment, although direct observations of sediment porewater concentrations would be needed to further evaluate this mechanism. It is likely that the sediment phosphorus peak corresponds to the peak in hatchery phosphorus load which occurred in the mid 1970's. Recycling from historical sediments would be expected to decline as the enriched sediments are depleted and/or buried by fresh sediment.

The quantity of phosphorus stored in surface sediments is large in relation to the quantity stored in the water column and in relation to the external load. Data from the 1988 core indicate that ~4000 mg/m^2 of phosphorus storage in the top 1-centimeter of sediment. The can be compared with ~62 mg/m^2 of storage in the water column at an average concentration of 8 ppb and with an average external loading rate of ~300 mg/m^2-yr in 1990-1996. The large storage of P in the sediment indicates that the lake may have a long "memory" of historical phosphorus loads.

Regardless of mechanism, recycling of sediment phosphorus would be expected to cause a lag in the lake response to reductions in external phosphorus load. Given the evidence of recycling and the substantial reductions in hatchery phosphorus load which occurred between 1977 and 1993, it is unlikely that the lake was equilibrium with external loads during 1993-1996. Further reductions in lake phosphorus concentrations and increases in transparency may occur under the same (1993-1996) external loading regime, as the effects of historical loads are diminished (either by flushing or burial).

Mass-Balance Modeling

This section describes the development, testing, and application of models for predicting the response of lake P concentrations and transparency to variations in phosphorus load on monthly and yearly time scales. The monthly model represents an ambitious attempt to simulate lake variations on a time scale that is generally not attempted in modeling responses to nutrient loads. The exercise is justified in this case because of the apparent sensitivity of lake transparency to phosphorus variations on monthly time scales (Figures 24-27) and because of the interest in evaluating effects of seasonal variations in phosphorus loads from the watershed and hatchery on lake responses. A second model uses a yearly time step for the phosphorus balance and probability models to represent seasonal variations in phosphorus concentrations and transparency. Although it provides less temporal resolution, the second model has fewer requirements in terms of input data, parameter estimates, and assumptions. Both models are used to evaluate impacts of alternative loading scenarios. Comparisons of results provide a basis for evaluating the sensitivity of predicted responses to alternative modeling approaches.

Monthly Model

A monthly phosphorus mass-balance model is coupled with empirical relationships between volume-weighted lake P concentrations and trophic response variables (chlorophyll-a and transparency). Data from 1990-1996 are used for calibration. The framework is tested by hind-casting historical conditions and comparing observed and predicted values using data from 1975-1989. Results are used to evaluate hatchery impacts under historical, current, and potential future loading regimes.

Phosphorus Balance

Coupled water-balance and mass-balance equations are used to predict lake volume-weighted-mean P concentrations on monthly time scale. Model structure is illustrated in Figure 35. Although simpler and parameterized differently, the model is similar to those developed by Lorenzen et al. (1976) for Lake Washington, Chapra (Chapra & Reckhow, 1983) for the Great Lakes, and Canale et al. (1991) for Platte Lake. The water column P balance is given by:

$$V \ dC/dt = W_e + W_r - Q\,C - U\ C\,A$$

where,

C = volume-weighted-mean concentration (ppb)
W_e = external loading (kg/yr)
W_r = sediment P recycle (kg/yr)
Q = outflow rate (hm^3/yr)
A = lake area (km^2)
U = gross settling rate from water column (m/yr)

The mass balance for labile sediment P is given by:

$$dM/dt = U\,C\ A - W_r - W_b = U\,C\,A - M / T_s$$

$$W_r = (1 - F_b)\,M / T_s$$

$$W_b = F_b\,M / T_s$$

where,

M = labile sediment P (kg)
W_b = sediment P burial (kg/yr)
T_s = sediment P residence time (years)
F_b = sediment P burial fraction (-)

The labile sediment component includes surface floc and sediments containing phosphorus subject to recycling induced by various physical, chemical, and biological mechanisms. As discussed above, important mechanisms in Platte Lake are likely to include decomposition of settling organic P, dissolution of calcite (releasing adsorbed P), wind-induced resuspension, and diffusion from bottom sediments (reflecting historical phosphorus loads). Direct measurements of labile sediment P and recycling rates are not available and would be difficult in any case. Sediment parameter estimates (T_s, F_b) are inferred from water-column concentration dynamics.

At steady-state (constant lake volume, external load and lake discharge), the following solutions are reached:

$$V\, dC / dt = W_e - Q\, C^* + (1 - F_b)\, M^* / T_s - U\, A\, C^* = 0$$

$$dM^*/dt = U\, C^*\, A - M^*/T_s = 0$$

$$C^* = W_e / [\, Q + U^*\, A]$$

$$U^* = F_b\, U$$

$$M^* = U\, C^* A\, T_s$$

where,

* = superscript denoting steady-state conditions

The steady-state model assumes that sediment feedback is in equilibrium with current external loads and is equivalent to that developed by Chapra (1996) for Platte Lake. Based upon data from 1991-1993, Chapra estimated a net settling rate (U^*) of 20.5 m/yr. Because of sediment feedback, the net settling rate will be less than U^* in years immediately following a reduction in external load and greater than U^* in years following an increase in external load. The rate of recovery of lake P concentration and net settling rate following a change in external load is determined by the P residence time:

$$T_p^* = (C^*V + M^*) / W_e$$

$$= (1 + U\, T_s / Z)\; C^*\, V / W_e$$

where,

T_p^* = Phosphorus residence time at steady-state (years)

Z = Mean depth (meters)

Solution of the above coupled differential equations representing the dynamic mass balance requires estimates of initial conditions for the state variables (C,M), driving variable time series (W_e, Q), and three parameters (U, T_s, and F_b). Model equations can be integrated analytically to predict the time series of lake P concentrations, given the time series of driving variables specified at any time step.

A sediment P residence time (T_s) of approximately 2 years is consistent with the observed recovery of the net settling rate in 1996 after reductions in load between 1993 and 1994 (Figures 32, 33) and the apparent time lag in historical transparency responses to reductions in hatchery load (Figure 29). The remaining P-balance parameters (F_b, U) are calibrated to predict 1990-1996 yearly-mean lake P concentrations in Figure 36. Resulting parameter estimates are F_b = 0.4, T_s = 2 years, and U = 45 m/yr. These values correspond to a steady-state settling velocity (U* = F_b U) of 18 m/yr. Figure 36(top) shows sensitivity to the sediment P burial fraction (F_b) over a range of 0.2 to 1 for a constant sediment P residence time (T_s) of 2 years. The Figure 36 (bottom) shows sensitivity to sediment P residence time over a range of 1 to 10 years for a constant burial fraction of 0.4. For each set of assumed F_b and T_s values, the gross settling rate (U) has been adjusted to give an unbiased prediction of the average phosphorus concentration over the entire 1990-1996 period (8.1 ppb).

Model predictions are relatively insensitive to calibrated values for F_b and $T_{s.}$ With a burial fraction of 1.0, the model is equivalent to the steady-state formulation and substantially over-predicts the sensitivity of lake P concentrations to reductions in external load between 1990 and 1996. With the calibrated parameters, the predicted response is buffered by sediment P recycling. The time scale of sediment P response (2 years) is short relative to that estimated in similar modeling efforts of other lakes (> 10 years, Lorenzen et al., 1976, Chapra & Reckhow, 1983). The short time scale for Platte Lake may reflect (a) relatively rapid burial and stabilization of newly-deposited sediment P as a result of the calcite precipitation process; and (b) apparent unimportance of redox-driven recycling mechanisms.

For all parameter sets, the model over-predicts the observed P concentration in 1991 and under-predicts the observed P concentration in 1995. As discussed above, the relatively low P concentration observed in 1991 was possibly related to strong thermal stratification and early warming of the lake in May, which resulted in an extended growing season. Under-prediction of the 1995 lake concentration is possibly related to the fact that tributary inflow concentrations were not monitored in that year. Uncertainty in 1995 load estimates is further increased by the fact that Platte River flow (Figure 2) dropped below the range values experienced during periods of stream concentration monitoring; load estimates for 1995 are thus based upon extrapolation of flow/concentration relationships derived from other years. The model provides an acceptable fit of data from the remaining years (1990, 1992-1994, & 1996).

As shown in Figures 32 and 34, net settling rates vary seasonally in Platte Lake. To reflect these variations, monthly scale factors are applied to the gross settling rate (U) when the model is run at a monthly time step. Twelve monthly scale factors have been calibrated so that the model gives unbiased predictions of the average observed P concentration in each month, based upon 1990-1996 data. Parameter estimates and model performance statistics are summarized in Table 4. Monthly scale factors range from 0.47 in November to 1.5 in August. This procedure ascribes seasonal variations to the gross settling rate. Seasonal variations in sediment P release rate (reflected in the F_b and T_s parameters) may also occur. It would be impossible, however, to calibrate a model with seasonal variations in both the net settling rate and sediment release rate based only upon observations of water-column P concentration. Further analysis indicates that assigning all of the seasonal variations to the sediment release rate yields results that are similar to those obtained assigning seasonal variations to the gross settling rate.

Figure 37 shows observed and predicted monthly-mean P concentrations in 1990-1996. As indicated in Table 4, the residual standard error is 1.2 ppb (adjusted for the 14 degrees of freedom used in calibrating the model). Residual variance reflects errors in the "observed" monthly mean concentrations calculated from 1-4 sampling dates per month, errors in the observed and estimated loads, as well as model errors. Based upon variance components in the 1990-1996 data set, the standard error of measured monthly-mean concentrations estimated from 1-4 sampling events per month ranges from 0.74 to 0.37 ppb. The year-to-year distribution of residuals is similar to that obtained when the model is applied at a yearly time step (Figure 36). The model over-predicts observed phosphorus concentrations in 1991 and under-predicts them from mid 1994 to early 1996. The later period corresponds well to the periods in which (a) river inflow concentrations were not measured (Figure 2) and (b) reported Platte River flows deviated significantly below those predicted from the Jordan River gauge (Appendix B, B-2). Errors in the watershed load estimates could account for the 1994-1996 deviations. Testing of the model against historical data is described below.

Figure 38 shows sensitivity to model parameters (T_s, F_b, U*) and to monthly scaling of the gross settling rate. Predictions are relatively insensitive to T_s and $F_{b,}$, but sensitive to net settling rate U* (= F_b U). Sensitivity to T_s has been tested over a wide range (1 to 10 years). Because sediment feedback has a stronger influence in periods following a change in external load, sensitivity to Ts is very low in 1990-1992, but increases after the reduction in hatchery load between 1992 and 1993. Monthly scaling of the gross settling rate is important to capture seasonal variations in lake P concentration.

<u>Chlorophyll-a & Transparency Models</u>

Table 4 summarizes model equations and coefficients for predicting chlorophyll-a and transparency as a function of volume-weighted-mean P concentration on a monthly time scale. These relationships are linked with the mass-balance model described above for predicting monthly lake P concentrations as a function of phosphorus loads and other driving variables.

The chlorophyll-a model (Figure 39) is derived from a regression analysis of monthly-mean chlorophyll-a concentration against monthly volume-weighted-mean P concentrations using data from 1994-1996, when a fluorometric procedure was employed for measuring chlorophyll-a. The regression has a logarithmic slope of 1.48, similar to that inherent in Carlson's Trophic State Index equations (1.47, Carlson, 1977). The intercept varies seasonally, with lower values in February-March, as compared with the remaining months. Figure 40 plots observed and predicted chlorophyll-a concentrations as a function of time. The model over-predicts observed values in 1990-1992, when a spectrophotometric procedure was used for measuring chlorophyll-a concentrations. As shown in Figure 39, deviations in the spectrophometric measurements tend to be stronger in the growing season than in the late winter and early spring. It is possible that these deviations reflect calcite interference in one or both of the analytical procedures.

Between May and September, transparencies are predicted as a function of volume-weighted-mean P concentration using the model shown in Figure 41. The model represents the threshold response of the extinction coefficient to increasing P concentration. The extinction coefficient increases at P concentrations above the calibrated threshold of ~ 7.5 ppb. Variations in the response reflect influences of temperature, stratification, and other random factors influencing algal growth and calicite precipitation. Model coefficients have been calibrated to data from 1990-1996, excluding June 1995. Factors contributing to the extreme whiting event in June 1995 are discussed above (see Lake Transparency Dynamics).

In the remaining months (October-April), transparency is predicted using a linear model which relates the light extinction coefficient to fluorometric chlorophyll-a concentration. Intercepts (non-algal light extinction coefficients) are adjusted seasonally (Figure 15) and have been calibrated to 1994-1996 data.

Observed and predicted monthly-mean Secchi depths for 1990-1996 are plotted in Figure 42. Predicted values in are based upon observed phosphorus concentrations. Predictions using phosphorus concentrations derived from the mass-balance model are discussed in the next section.

<u>Model Testing</u>

The monthly model has been tested by comparing observed and predicted values for 1975-1989 (prior to the 1990-1996 calibration period). Figure 43 shows observed and predicted values for phosphorus and Secchi depth. Figure 44 shows observed and predicted values for chlorophyll-a and lay Secchi depth. Residuals (observed – predicted values) are plotted against year in Figure 45. Residual standard errors for the model calibration (1990-1996) and model testing (1975-1989) periods are listed in Table 4.

Simulations of pre-1990 conditions are limited by the fact that Platte River inflows and loads were not directly measured, but have been estimated using procedures described in Appendix B. Historical lake phosphorus measurements do not have the vertical, temporal, or analytical resolutions characteristic of the 1990-1996 data. Given the low range of phosphorus values in Platte Lake, differences in sampling and analytical methods could easily introduce an important variance component. The above factors contribute to a higher residual standard error for phosphorus in 1975-1989 (2.9 ppb), as compared with 1990-1996 (1.2 ppb). Despite these limitations, the model generally gives unbiased predictions of historical phosphorus concentrations and accurately hindcasts concentrations measured during relatively intensive surveys conducted in 1981-1982 (Clean Lakes Study, MDNR, 1983) and 1988. It over-predicts measured concentrations in 1980 and 1987.

Residual standard errors for hindcasts of historical Secchi depths are lower than those calculated for the model calibration period, both for CM/STORET measurements (2.3 vs. 2.8 ft) and for Lay measurements (2.4 vs. 3.9 ft). This may reflect the lower range of the historical transparency measurements and a dependence of model error on measurement scale (fixed percentage error vs. fixed absolute error). In any case, the model hindcasts most of the historical transparency measurements compiled from STORET (Figure 43). The divergence noted for phosphorus in 1980 and 1987 does not appear in the transparency time series.

The model tends to over-predict transparency measurements collected under the lay monitoring program (Figure 44). This result is expected based upon the apparent differences between Lay and CM transparency measurements in 1990-1996 (Figure 28). The dashed line in Figure 44 shows predicted lay transparency values adjusted for measurement bias using the regression model in Figure 28. The applicability of the regression to historical data is unknown, however. Systematic differences between lay and STORET transparency measurements are not apparent during the period of the Clean Lakes study (1981-1982), when transparencies were in a relatively low range.

Interpretation of chlorophyll-a simulations is complicated by measurement biases resulting from differences in analytical procedure. The model has been calibrated to fluorometric measurements in 1994-1996 and under-predicts spectrophometric measurements in 1990-1992. Information on historical analytical procedures could not be located in published reports. Observed and predicted ranges are similar in 1976-1977 and 1985-1988. Chlorophyll-a measurements from the Clean Lakes study (1981-1982) are over-predicted. Secchi depths (computed from predicted chlorophyll-a concentrations) are accurately simulated during this period, however (Figure 43). Because of biases resulting from analytical procedures, chlorophyll-a residuals are not shown in Figures 45.

The model over-predicts seasonal minimum transparencies in 1992, 1994, and 1995. This is expected based upon the fact that the model does not incorporate effects of temperature and thermal stratification , which amplified whiting events in those years (see Lake Transparency Dynamics). Deterministic simulation of these factors would involve a much more complicated model. A stochastic approach to predicting transparency response is described in the next section.

Over-prediction of transparency in 1995 is related to under-prediction of P concentrations in that year and to the fact that temperature and stratification effects are not built into the model. As discussed above, under-prediction of 1995 phosphorus measurements is related to a low apparent settling rate (Figure 33) and/or lack of errors in the watershed loads estimates resulting from the lack of stream sampling during that year.

Figure 46 shows long-term variations in the phosphorus fluxes driving the predicted lake concentration time series. Plotted terms include total external load, hatchery load, and sediment recycle. The time lag in the response of the recycle term to changes in external load is readily apparent. Future reductions in sediment recycle and lake concentrations are expected if future external loads remain at or below 1993-1996 levels. Based upon the calibrated parameter values and equations presented above, phosphorus residence time in the lake (including both water column and labile sediment components) is approximately 2.9 years under 1993-1996 conditions. This is considerably longer than the residence time in the water column computed from external load and average water-column concentration (2.8 months). Because of the time lag in sediment response, the predicted average concentration in 1993-1996 (7.4 ppb) is higher than the predicted steady-state concentration in response to 1993-1996 loads (7.0 ppb). This comparison demonstrates the approximate impacts of sediment P recycling on current P levels.

<u>Simulation of Alternative Phosphorus Loading Scenarios</u>

This section describes application of the above model to evaluate the impacts of phosphorus loads from the Platte River hatchery and watershed on lake P concentrations and Secchi depths. The following hatchery load scenarios have been evaluated:

1. Historical (1975-1996)
2. 1975-1981
3. 1982-1988
4. 1989-1992
5. 1993-1996
6. Zero Loads
7. 1993-1996, with watershed loads adjusted for trends
8. Zero Loads, “ “ “ “ “

To reflect hydrologic variations, each scenario has been run for the 1975-1996 period using input values listed in Appendix G. Phosphorus loads from sources other than the watershed and hatchery have been estimated using the same assumptions used to derive lake mass balances for 1990-1996. Initial conditions in the water-column and sediment have been established by running the model for two consecutive 1975-1996 sequences. Results from the second sequence are presented. Scenarios 2-5 reflect the four major loading regimes that occurred between 1975 and 1996 (Figures 29 & 31). Temporal variations in hatchery loads under each scenario are represented by repeating each monthly load sequence for the entire 1975-1996 period. For example, Scenario 5 involves 5.5 consecutive 1993-1996 load sequences imposed on the 1975-1996 hydrologic record. Scenario 6 represents a base condition with historical watershed loads. Hatchery impacts under Scenarios 1-5 are evaluated by comparing results with Scenario 6. Scenario 5 represents a projection of lake conditions under current (1993-1996) loading regimes.

Projections of future concentrations relative to the 8 ppb goal are extremely sensitive to the assumed phosphorus loads from sources other than the hatchery, which contributed more than 96% of the total estimated load between 1993 and 1996. Future increases or decreases in loads could result from watershed development and/or management could have a large effect on the feasibility of achieving the 8 ppb goal, regardless of its interpretation and regardless of hatchery load. As noted above (see Watershed Data Analysis), decreasing trends in phosphorus concentrations from Brundage Creek and the North Branch of the Platte River were detected in the watershed monitoring data. Because of these decreasing trends, future watershed loads may be lower than those assumed in Scenarios 1-6. Scenario 7 has been constructed to reflect 1993-1996 hatchery loads with watershed loads adjusted to 1996 conditions using trend magnitudes estimated from historical stream concentration data, as described above (see Watershed Data Analysis). If the apparent trends in stream concentration data represent long-term changes, then Scenario 7 represents the best projection of future lake conditions with hatchery loads at 1993-1996 levels. Scenario 8 uses de-trended watershed loads with zero hatchery load. Scenarios 7 and 8 also demonstrate the sensitivity of the simulations to small reductions in phosphorus loads from sources other than the hatchery, which may be achievable via watershed management or other control measures.

Changes in analytical procedure complicate the interpretation of chlorophyll-a results. Algal blooms (typically defined as chlorophyll-a concentrations exceeding 10 ppb, Walmsley 1984; Walker, 1985b) have generally not been observed in Platte Lake and are not predicted under any loading scenario. It is assumed that chlorophyll-a predictions are only relevant insofar as they influence predicted transparencies. For this reason, chlorophyll-a results are tabulated but not discussed.

When interpreting model results, it is important to consider that predictions of relative impact (comparisons with Scenario 6) are more reliable than predictions of absolute values (concentrations or transparencies) and exceedance frequencies (phosphorus > 8 ppb or Secchi depth < 10 ft). Relative predictions are generally less sensitive to model input variables and parameter estimates. This is true of modeling results in general. For example, a small percentage shift in future phosphorus loads from the watershed and other non-hatchery sources could have a major impact on the predicted 8-ppb exceedance frequencies but would have very little impact on the relative predictions of hatchery impact. This sensitivity is demonstrated in results from Scenarios 7 and 8.

Table 5 summarizes the frequency distributions of monthly hatchery loads, P concentrations, and Secchi depths under each scenario. Phosphorus distributions are represented by the mean, minimum, and maximum monthly values and by the frequency of months with concentrations exceeding 8 ppb. Secchi distributions are represented by the mean, minimum, and maximum monthly values and by the frequency of months with values less than 10 feet. The mean, minimum, and maximum impacts on monthly phosphorus concentrations and transparencies are also presented.

Table 6 summarizes results in a similar fashion based upon predicted values for the growing/recreational season (May-September), which are relevant for evaluating phosphorus impacts on eutrophication and related water uses. Two sets of May-September Secchi frequencies are presented for each Scenario. One is based upon the model calibration to Court Master's Secchi measurements. The other is based upon Lay Secchi measurements. As indicated in Figure 28, the latter measurements tend to be higher when the data are paired by month. Based upon the regression equation in Figure 28, a Lay Secchi depth of 10 feet is approximately equivalent to a CM Secchi depth of 7.3 feet. Thus, the frequency of CM values less than 7.3 feet estimates the frequency of Lay measurements less than 10 feet. Results derived from Table 6 are displayed in Figure 47 (phosphorus distributions), Figure 48 (Secchi distributions), and Figure 49 (phosphorus & Secchi frequencies). May-September results are emphasized in the discussion below.

Average hatchery impacts on May-September lake P concentrations decrease from 4.0 ppb under the 1975-1981 loading regime to 0.24 ppb under the 1993-1996 loading regime. Corresponding maximum monthly impacts are 5.1 and 0.46 ppb, respectively. Expressed as a percentage of the baseline condition (Scenario 6), the average impact of Scenario 5 on annual lake P concentrations (0.28 / 7.58 = 3.7%, Table 5) is consistent with relative loads from the hatchery and other sources (96/2609 kg/yr = 3.8%). Average and maximum impacts on transparency decrease from 5.2 feet and 9.2 feet, respectively, under 1975-1981 loads to 0.24 feet and 1.09 feet, respectively, under 1993-1996 loads.

Monthly simulation results for phosphorus and transparency are plotted in Figures 50 and 51, respectively. To simplify the plots, only results from Scenarios 1, 5, and 6 are shown. Annual mean and May-September mean P concentrations are plotted in Figure 52. Because of sediment P recycling (which reflects antecedent loads), predicted P concentrations and hatchery impacts under Scenario 1 between 1993 and 1996 are higher than those predicted under Scenario 5.

Baseline conditions and impacts of 1993-1996 hatchery loads are plotted as a function of month in Figure 53 (P concentrations), Figure 54 (Court Master's Secchi depth), and Figure 55 (Lay Secchi depth). These are based upon comparisons of Scenarios 7 and 8, which represent the best projections of future conditions with and without hatchery loads at 1993-1996 levels. Predicted frequencies of phosphorus concentrations exceeding 8 ppb and Secchi depths less than 10 feet have been calculated separately for each month and scenario, based upon results of the 22-year simulations.

Annual Model

In order to simulate lake phosphorus variations with a monthly time step, the model described above has relatively high requirements for input data (e.g., monthly flows and loads from each source) and parameter estimates (12 monthly scaling factors for net settling rate). In particular, it relies on estimates of monthly watershed flows and loads for 1975-1996 developed from limited watershed monitoring data (flows from 1990-1996 and stream concentrations primarily from January 1990-May 1994 and March-December 1996). This section describes an alternative modeling approach which has less demanding input and parameter requirements. Phosphorus balances are simulated with a yearly time step. Stochastic (vs. deterministic) methods are used to predict phosphorus and transparency frequency distributions.

The phosphorus balance is formulated on a yearly time step using the same equations developed for the monthly model. Given relatively limited watershed monitoring data, the model is formulated to simulate lake responses to yearly variations in hatchery phosphorus loads with phosphorus loads from other sources held constant at average 1990-1996 values. Effects of year-to-year variations in watershed loads are reflected in a stochastic term represented by a lognormal frequency distribution with scale calibrated to 1990-1996 data. Similarly, within-year (daily) variations in phosphorus concentration are represented by a lognormal frequency distribution superimposed on the predicted yearly mean concentration. Explicit consideration of variance terms permits estimation of 8-ppb exceedance frequencies on yearly and daily time scales.

The following equations describe the predicted frequency distributions of yearly and daily volume-weighted-mean concentrations developed from the mass balance equation:

$$P_y = P_m \exp(F_y)$$

$$P_d = P_m \exp(F_y + F_d)$$

where,

P_m = mean concentration predicted by mass-balance model (ppb)

P_y = yearly mean concentration (ppb)

P_d = daily (sampling event) P concentration (ppb)

F_y = term representing random year-to-year variations in P concentration normal deviate with mean = 0.0 and standard deviation = 0.057

F_d = term representing random within-year variations (among sampling events) normal deviate with mean = -0.02 and standard deviation = 0.190

The predicted mean concentration time series (P_m) has been generated by integrating the mass-balance model at a yearly time step using parameter estimates (U, F_b, & T_s) developed above for the monthly model and the reported yearly-mean hatchery loads.

The year-to-year variation term (F_y, mean = 0.0 and standard deviation = 0.057) has been calibrated to the error distribution of residuals (observed – predicted yearly-mean concentrations in 1990-1996, Figure 56). The F_y term reflects year-to-year variations in watershed loads and other random factors that influence yearly mean lake P concentrations (exclusive of hatchery loads, which are reflected in the P_m time series).

The within-year variation term (F_d, mean = -0.02, standard deviation = 0.190) has been calibrated to the distribution of observed volume-weighted-mean concentrations with year-to-year variations removed. This distribution is characterized by the mean and standard deviation of the following statistic computed across all 1990-1996 sampling events:

$$x = \ln (C_{ey} / C_y)$$

C_y = observed volume-weighted-mean concentration in year y (ppb)

C_{ey} = observed volume-weighted-mean concentration on event e in year y.

Table 7 lists model input and output variables for simulation of the 1975-1996 period. Figure 56 compares observed and predicted yearly volume weighted-mean concentrations. Frequency distributions of predicted yearly-mean concentrations are represented by the 10^{th}, 50^{th}, and 90^{th} percentiles. As observed in monthly simulations (Figures 43 & 45), residual variance is considerably higher for the historical period (1975-1989), as compared with the calibration period (1990-1996). Aside from limitations resulting from low analytical, vertical, and temporal resolution of historical data, observed annual volume-weighted-mean concentrations are relatively uncertain and potentially biased because of incomplete seasonal coverage (all months were not sampled). With the exception of two years (1980 and 1987), the model provides reasonable estimates of historical spring overturn (April) concentrations. Phosphorus concentrations in 1980 and 1987 were also over-predicted by the monthly model (Figure 43).

A logistic model (Mosteller & Tukey, 1977) is used to simulate transparency responses to variations in lake phosphorus concentrations predicted by mass-balance:

$$F_c = 1 / [1 + \exp (-(P_m - m) / s)]$$

where,

S_c = Secchi depth criterion (6, 8, 10, or 12 feet)
F_c = Frequency of May-September Secchi depths below S_c
P_m = Predicted mean P concentration (ppb)
m,s = empirical coefficients (ppb)

The empirical coefficients (m,s) represent the midpoint and spread of the transparency response. When $P_m = m$, the predicted frequency is 50%. When $P_m <$ m, the predicted frequency is less than 50%, depending on the magnitude of the parameter s. While the monthly model attempts to predict transparency variations deterministically as a function of monthly volume-weighted-mean concentration, the logistic model uses a stochastic approach which simulates the probability or risk of Secchi values below a given criterion as a function of yearly mean phosphorus concentration. The risk reflects the combined effects of seasonal variations in phosphorus concentration around each annual mean and the resulting transparency response. Average effects of water temperature and other controlling factors are embodied in that risk. Year-to-year in such factors are reflected in the model error term. The parameter m can be interpreted to reflect the instantaneous phosphorus concentration corresponding to an average 50% risk of observing Secchi depths below the criterion.

For each Secchi criterion, the empirical coefficients m and s have been calibrated to historical lay transparency data (Figure 30) using a linear regression model of the following form:

$$\text{Logit}(F_c) = \ln [F_c / (1 - F_c)] = a + b P_m + e$$

$$m = - a / b$$

$$s = 1 / b$$

where,

a = regression intercept

b = regression slope

e = error term

Regression analyses provide estimates of a, b, and e for each Secchi criterion. Calibrations are based upon data from years with direct observations of hatchery phosphorus loads and transparency (1980-1996).

With a weekly monitoring program consisting of 21 observations between May and September, the minimum detectable frequency (MDF) for any Secchi criterion is 1/21 or 4.8%. In a year when all measured values are above 10 feet, the true frequency of values <10 feet would be less than the MDF (0-4.8%). Similarly, in a year when all measured values are below 10 feet, the true frequency of values <10 feet would be greater than 100-MDF (95.2 -100%). To reflect constraints imposed by the number of sampling dates and to allow calibration of the logistic response models (which not allow observed frequencies of 0 or 100%), observed frequencies have been assigned a value of MDF/2 (2.4%) in years when all values were above the criterion and a value of 100 - MDF/2 (97.6%) in years when all values were below the criterion.

Figure 57 shows observed and predicted Secchi interval frequencies as a function predicted lake P concentration and year. Model predictions are driven exclusively by variations in hatchery load and successfully capture long-term transparency responses. The most pronounced response was the sharp reduction in the frequency of values less than 8 or 10 feet following the load reduction in 1982. Observed and predicted frequencies less than 12 and 10 feet gradually drifted downward during the 1980's.

Effects of year-to-year variations in watershed loads and other factors influencing the transparency response to a given hatchery load are reflected by the differences between observed and predicted frequencies in each year. Temperature and stratification effects discussed above (see Lake Transparency Dynamics) are probably responsible for the fact that observed Secchi frequencies were greater than predicted values in 1995 and lower than predicted values in 1991. The transparency response does not seem to be sensitive to wet vs. dry year variations. For example, even though the observed lake phosphorus concentration exceeded the predicted value in 1990 (a result of high runoff loads in that year), observed and predicted Secchi interval frequencies were similar in that year.

Results of scenario evaluations using the annual model are summarized in Figure 58 and Table 8. Steady-state responses are predicted for the same scenarios evaluated above with the monthly model. Results are generally consistent with those obtained from the monthly model. Most of the benefits potentially derived from reductions in hatchery load are reflected in the 1993-1996 scenarios. Long-term average P concentrations range from 6.8 to 7.6 ppb for Scenarios 5-8 and are identical to those predicted by the monthly model. Frequencies of annual mean concentrations exceeding 8 ppb under Scenarios 5-8 range from 17% to 2% based upon the annual model, as compared with 23% to 0% based upon the monthly model. For the same scenarios, predicted frequencies of lay transparency measurements less than 10 feet range from 8 to 3%, as compared with 8% to 2%. The validity of the results is supported by the consistency between the predictions of the monthly and yearly models, which have been developed using different assumptions, data sets, and modeling approaches.

Discussion

Results Relative to the 8 ppb Goal

Based upon data analyses and modeling results presented above, this section discusses observed and predicted lake conditions in relation to the 8 ppb goal established by Court Order (Brown, 1988ab). The feasibility of attaining the goal under any loading scenario is largely dependent upon whether the goal is interpreted as a target (central tendency for future lake measurements) or as a limit (value not to be exceeded) and upon the time scale and season over which the goal is assumed to apply. The Court Order does not specify the applicable time scale, season, or exceedance frequency that would constitute non-compliance. Alternative interpretations of these parameters can have significantly different management implications.

Modeling results provide a basis for evaluating hatchery load impacts under consistent hydrologic and watershed loading regimes. The following discussion focuses on results of the monthly mass-balance model, which are reasonably consistent with those of the annual model. Table 9 summarizes predicted 8 ppb exceedance frequencies for various scenarios and time scales. Scenario 7 represents the best projection of future conditions with hatchery loads at 1993-1996 levels and watershed loads at current levels. Scenario 8 represents the same conditions without hatchery loads.

<u>Interpretation of 8 ppb as a Limit on the Long-Term Average</u>

If the 8-ppb goal is interpreted as a target (central tendency for lake measurements) or as a limit on the long-term annual average or long-term growing-season average, then model results indicate that the goal would be achieved under both current (7.6 ppb annual average, 7.2 ppb May-September average) and zero-load scenarios (7.3 ppb and 6.9 ppb, respectively). Lower values are predicted when de-trended watershed loads are used (Scenarios 7 & 8). Model results are supported by the fact that the observed annual-average and May-September average concentrations were significantly below 8 ppb in 1993-1996 (Tables 1 & 2).

Predicted long-term-average phosphorus concentrations for Scenarios 5-8 range from 6.8 to 7.6 ppb. These are based upon a net setting rate of 18 m/yr calibrated to 1990-1996 data. Chapra (1996) calibrated his model to 1991-1993 data and obtained a net settling rate of 20.5 m/yr. With Chapra's settling rate, predicted long-term-average phosphorus concentrations for Scenarios 5-8 would range from 6.3 to 7.1 ppb.

Canale et al. (1991) developed a coupled water-column and sediment phosphorus model for Platte Lake and used it to evaluate alternative loading scenarios. With a 90% reduction in hatchery load, their model predicted long-term-average phosphorus concentrations ranging from 7.2 to 7.6 ppb, depending upon extent of watershed development. They concluded that this plan was "sufficient to meet the water quality standards mandated by the court". Using the same interpretation, results for Scenarios 5-8 (6.8 to 7.6 ppb) would also be sufficient.

<u>Interpretation of 8 ppb as a Limit on the Annual Average</u>

If the 8 ppb goal is interpreted as a limit on each annual average, then model results indicate that the limit would be exceeded in 23%, 9%, 0%, and 0% of future years under Scenarios 5, 6, 7, and 8, respectively. If the goal is assumed to apply to each May-September average (more relevant than the annual mean as an indicator of phosphorus impacts on algal growth, transparency, and recreational uses), then the limit would be exceeded in 9%, 0%, 0%, and 0% of future years, respectively. Because the simulations employ a 22-year period of record, a predicted exceedance frequency of 0% should be interpreted to represent an actual frequency less than ~5% (1/22 years).

Results for Scenarios 7 and 8 are compatible with an 8 ppb limit for annual averages average. Results for Scenarios 5 and 6 could also be interpreted as compatible, depending upon whether allowance is made for exceedances in high-runoff years. Fuss(1989) pointed out the likelihood that lake concentrations would exceed 8 ppb during periods of high runoff without the hatchery discharge and that year-to-year variations of ±15% (6.8 to 9.2 ppb) would be expected, based upon data from other lakes (Smeltzer et al, 1989).

Under Scenarios 5 and 6, exceedances of 8 ppb in the yearly or seasonal means are attributed primarily to high runoff from the watershed in specific years (e.g., 1990, 1979). Under Scenario 5, the predicted maximum annual-average concentration is 8.5 ppb and the predicted maximum May-September average concentration is 8.1 ppb (Table 6, Figure 47). These values are well within the expected 6.8 to 9.2 ppb range discussed by Fuss (1989). In each case, the increment above 8 ppb (0.5 and 0.1 ppb, respectively) is not measurable in the context of an intensive monitoring program similar to that conducted in 1990-1996. As indicated above (see Analyses of Variance), the standard error of the measured annual mean phosphorus concentration is typically 0.29 ppb and the 90% confidence interval for the true annual mean is typically ± 0.5 ppb of the measured mean. In order to state with confidence that the true annual mean is above 8 ppb, the measured annual mean would have to exceed 8.5 ppb. Predicted yearly means and May-September means are consistently below 8.5 ppb under Scenarios 5-6 and below 8.0 ppb under Scenarios 7-8.

Interpretation of 8 ppb as Target for Spring Overturn Concentrations

Based upon review of spring overturn phosphorus and summer water quality data from Platte Lake and other high-quality Michigan lakes, MDNR(1986) identified a number of eutrophication-related water-quality goals for Platte Lake and estimated that "to achieve these goals it would be necessary to reduce in-lake phosphorus concentrations to between 7 and 9 ug/L." MDNR further recommended "that Platte Lake be managed to achieve 8 ppb total phosphorus at spring overturn in most years". The above analyses of 1990-1996 monitoring data indicate that spring overturn P concentrations are less useful than May-September average P concentrations as predictors of summer transparency. Nevertheless, it seems appropriate to compare model results with the spring overturn criterion recommended by MDNR.

Figure 59 shows predicted phosphorus concentrations in April of each year between 1975 and 1996 for loading scenarios 1, 5, 6, 7, and 8. Predicted concentrations exceed 8 ppb in 77%, 64%, 36%, 23%, and 14% of the years, respectively. Depending upon interpretation of the phrase "in most years", Scenarios 6-8 could be viewed as consistent with MDNR recommendations, particularly if allowances are made for exceedences in high-runoff periods. Predicted values exceed the recommended 7 to 9 ppb range in two years (1979,1981) under all scenarios. A higher exceedance frequency is indicated in earlier portions of the 1975-1996 simulation period for all scenarios. This result can be traced to higher runoff flows and concentrations in March and April, as reflected in the total inflow load time series (Figure 46). Runoff flows and concentrations for 1975-1989 have been estimated based upon correlations between Platte River flows and Jordan River flows calibrated to 1990-1996 data (see Watershed Data Analysis & Appendix B). The predicted spring runoff flows in many of the early years in the simulation exceeded the range of values used in calibrating the flow- estimation procedure. It is possible that runoff flows and loads are over-estimated in earlier portions of the record. Exceedance frequencies in the last half of the simulation period, in which watershed flows and loads are based more upon direct monitoring data, are 55%, 45%, 18%, <5%, and <5% for Scenarios 1, 5, 6, 7, and 8, respectively. Future investigations of alternative procedures for estimating historical flows and loads might improve the accuracy of the simulations. Based upon these results, it is reasonable to conclude that Scenarios 6-8 are consistent with MDNR recommendations.

Interpretation of 8 ppb as a Limit on Monthly or Daily Averages

If the 8 ppb goal is interpreted as a limit on each monthly average, the limit would be exceeded in 38%, 28%, 22%, and 15% of future months under Scenarios 5, 6, 7, and 8, respectively. Results for May-September are 32%, 26%, 18%, and 10%, respectively. It is unlikely that exceedance frequencies approaching 0% could be achieved on a monthly (or daily) basis because of natural seasonal and year-to-year variations in the watershed and lake. Fuss(1989) recommended that compliance be determined based upon a maximum 8-ppb exceedence frequency of 20% in samples collected from the euphotic zone during the ice-free season. Interpretation of the 8 ppb goal as a number never to be exceeded on a daily or monthly basis is not realistic in the context of inherent natural variability. For this reason, expression of phosphorus goals on a seasonal-average basis has been recommended in a lake management context (North American Lake Management Society, 1990; Walker, 1985b).

Simulations indicate that if the goal is interpreted as a limit on monthly or daily averages, it would not be attainable via reductions in hatchery loads alone. These results supported by Chapra's (1990) modeling results for Platte Lake, which indicated that average P concentrations below 6.9 and 6.0 ppb would be required to maintain lake concentrations below 8 ppb more than 75% and 90% of the time, respectively. Chapra's model indicated that achieving these concentration levels would require reductions of 6 to 18% in non-hatchery sources, in addition to complete elimination of the hatchery load.

Figure 60 plots 8-ppb exceedence frequencies against spring phosphorus concentrations using data from Platte and other high-quality Michigan Lakes. The data summary (Table 10) was used by Wandell et al. (1989) in discussing water-quality objectives for Platte Lake. Exceedance frequencies in Platte Lake have been calculated for the April-October period, when most of the other lake data were collected. This comparison demonstrates that variations in Platte Lake are not unusual in the context of other regional lakes and concentrations would be expected to exceed 8 ppb more than 20% of the time in the majority of lakes.

Figure 61 shows the relationship between measured annual-mean P concentrations and frequencies of daily means and sample values exceeding 8 ppb based upon Platte Lake monitoring data from 1990-1996. Exceedance frequencies have been calculated on a sample basis (all seasons and depths), seasonal basis (all samples collected between April and October at depths < = 30 feet), and daily basis (volume-weighted means on each sampling date, all seasons and depths). Regardless of how the frequencies are calculated, it is apparent that exceedance frequencies ranging from 40 to 60% are expected in years when the annual mean equals 8 ppb. An exceedance frequency of 20% was approached only in 1996, when the annual mean concentration was 7.3 ppb.

As a consequence of seasonal variations in load and settling rates, predicted 8-ppb exceedance frequencies and hatchery impacts vary with month. Figure 53 shows the seasonal distribution of average phosphorus concentrations, 8-ppb exceedance frequencies, and hatchery impacts based upon comparisons of Scenarios 7 and 8. Under both scenarios, exceedances of 8 ppb occur primarily in the Spring (March-June) and November. Relatively high concentrations and frequencies (>80% under both scenarios) in November reflect the low apparent settling rate in that month and are unlikely to have significant impacts because they occur outside of the growing season. Exceedances in the late spring (May & June, in particular) are potentially of greater concern because transparency is apparently sensitive to phosphorus variations during this period (see Lake Transparency Dynamics). The 8-ppb exceedence frequency between April and June is 21% under Scenario 8 (without hatchery loads) and 38% under Scenario 7 (with hatchery loads). Hatchery impacts appear larger when viewed in this context, but the magnitude of the impact in these months (averaging 0.3 ppb) is small in relation to the average concentration without the hatchery load (7.5 ppb, Figure 53). Predicted average concentrations are below 8 ppb in both cases. Higher phosphorus concentrations in these months reflect higher runoff concentrations and loads, as well as the timing of hatchery loads (Figure 32).

Results Relative to Transparency Goals

The 8-ppb phosphorus goal was established based partially concerns about impacts on water transparency, particularly relative to a 10-ft benchmark measured in 1940. On the scale of the Court Master's measurements, predicted transparencies would exceed 10 feet between 68% and 90% of months between May and September under Scenarios 5-8. On the scale of lay transparency measurements, transparencies would exceed 10 feet between 92% and 98% of months. Similar predictions are derived from the annual model (Figure 58, Table 8). These predictions are in sharp contrast to those observed during the period of peak hatchery loading (1977-1981), when lay Secchi measurements exceeded 10 feet only 4% of the time.

Substantial improvements in transparency have resulted from historical reductions in hatchery load (Figures 29, 31, 48, 49, 58). Figures 31 and 58 suggest that most of the benefits potentially resulting from control of hatchery loads were achieved by load reductions occurring prior to 1990. Decreasing trends in May-September transparencies were not detected between 1990 and 1996, despite the fact that hatchery loads were reduced by 69% between 1990-1992 and 1993-1996 (Figure 31). Further improvements in transparency potentially resulting from reductions in hatchery load below 1993-1996 levels may be difficult to detect in the presence of natural variations. Impacts of 1993-1996 hatchery loads on water transparency between May and September average –0.24 feet and range from 0 to –1.1 feet.

Because of the seasonal distributions of watershed and hatchery loads, average impacts on transparency in May and June (0.5 to 0.6 feet) are greater than those predicted for other months (<0.3 feet) (Figures 54 & 54). May-June impacts are relative to average transparencies of 10 feet based upon the Court Master's Secchi measurements (Figure 54) and 13 feet based upon lay Secchi measurements (Figure 55). The estimated impacts are small in the context of (a) typical seasonal and year-to-year variability in transparency (7 to 14 feet, Figure 12); (b) typical resolution of Secchi depth measurements (values are reported to nearest foot under the lay monitoring program and nearest 0.5 foot under the Court Master's program); and (c) random and systematic differences between the Court Master's and Lay transparency measurements made during the same periods (standard error = 1.7 feet, Figure 28).

Management Strategies

The above discussion indicates that the 8-ppb lake goal may be achievable under 1993-1996 hatchery loads, depending upon how the goal is interpreted. In the context of historical and recent watershed loads, 1993-1996 hatchery loads are consistent with annual and growing-season averages less than 8 ppb in most years. If the goal is interpreted as a target (central tendency for future lake measurements) or as a limit on annual, May-September, or spring overturn values, then it is likely to be attainable with hatchery loads at 1993-1996 levels. If the goal is interpreted as a limit on sample values, daily averages, or monthly averages, then it is not likely to be attainable under any of the scenarios evaluated (including complete elimination of hatchery loads). Recommended management strategies under each of these interpretations are discussed below.

Regardless of goal interpretations, historical discharges from the hatchery have apparently had significant impacts on lake phosphorus concentrations and transparency. Increases in transparency occurred following substantial reductions in hatchery load between 1981 and 1993. The lake may continue to improve under current conditions as effects of historical phosphorus loads recycled from lake bottom sediments diminish. These benefits appear to justify efforts to reduce hatchery loads.

Consideration should be given to revising the current hatchery discharge permit to ensure that loads do not increase above 1993-1996 levels. Given the nominal residence time of phosphorus in the lake water column (~2.8 months), limits on quarterly and annual loads are recommended to minimize lake impacts resulting from short-term and annual variations in load. Limits could be expressed in terms of maximum 3-month and 12-month rolling values. Between 1993 and 1996, the maximum 3-month rolling load was 55 kg and the maximum 12-month rolling load was 133 kg. Assuming that future variability in the hatchery discharge is similar to that experienced historically, operating within the above limits would ensure that average and maximum loads during the permit interval do not increase above 1993-1996 levels. Since they are based upon historical performance, these limits are technically achievable. With refinements in technology resulting from future research, lower limits may be achievable and justified in future permit intervals.

If the discharge limit were derived based upon the annual model (Table 8) with the objective of maintaining a long-term-average lake concentration < 8 ppb, then the limit would range from 240 to 420 kg/yr, depending upon whether historical (Scenario 6) or detrended (Scenario 8) watershed loads are assumed. The proposed 133 kg/yr limit based upon historical performance is considerably more stringent and would provide greater lake protection. This limit compares with the maximum historical load of ~1700 kg/yr (1976, Table 7) and with load estimates for the 1929-1969 period (prior to the hatchery expansion) ranging from 105 kg/yr (Canale et al., 1991) to 511 kg/yr (Grant, 1979). If an average estimate is taken for pre-1970 loadings, the proposed limit would more than offset any lake impacts resulting from expansion of the hatchery in the 1970's.

If the hatchery operates within the above suggested permit limits and if phosphorus loads from other sources do not increase, then modeling results and recent monitoring data indicate that annual-average, growing-season-average, and spring overturn phosphorus concentrations should be below 8 ppb in most years. If alternative interpretations of the goal indicate that further reductions are required (down to and including zero net load), then measures for achieving such reductions should be identified and evaluated from a cost-effectiveness standpoint.

With further improvements in hatchery operation, reductions in load below 1993-1996 levels may be achievable without substantial capital costs. Potential additional measures might involve modifications to the hatchery design and/or implementation of watershed nonpoint-source controls to offset hatchery loads. Use of non-point-source controls to offset point-source impacts has been shown to be a cost-effective means of achieving goals in other lake-management situations (Duda et al., 1988). Although opportunities for implementing non-point controls may be limited because a large portion of the Platte Lake watershed is undeveloped, such opportunities have not, to the author's knowledge, been systematically evaluated.

Benefits to the lake resulting from additional investments in hatchery or watershed controls can be expressed in terms of net reduction in phosphorus load. Removing the last few percent of the phosphorus load from any facility is typically very expensive on a $/kg-removed basis. Because the technologies for accomplishing this are often experimental and unproven, such measures typically involve large investments and high risk. Significant negative impacts on the pubic benefits and net economic viability of the facility may result. On the other hand, removing the first few percent of the load from a point or non-point source is typically relatively inexpensive on a $/kg-removed basis and may involve less risk because it can be accomplished with proven technology. For these reasons, evaluating control alternatives in the context of the entire watershed can produce management plans which accomplish a given load reduction with minimum investment and risk.

Under the 1993-1996 loading scenario, lake phosphorus concentrations are more likely to exceed 8 ppb on a monthly basis during periods of high watershed runoff. It may not be cost-effective to invest in further hatchery controls in an attempt to reduce loads during such periods, when relative contributions from the hatchery are very small (< 4 %). Identification of watershed source areas and evaluation of potential watershed controls (largely ignored under the 1990-1996 monitoring program) would be necessary to develop a cost-effective strategy for achieving further reductions in total loads and lake concentrations.

Figure 62 shows regional variations in phosphorus export from Platte River subwatersheds monitored under 1980-1981 Clean Lakes Study (MDNR, 1983). Export rates range from 0.4 to 23 kg/km^2-yr. There may be opportunities to achieve reductions in load from those areas which have relatively high export rates, including developed areas and roadways around the lake. At a typical rate of 60 kg/km^2-yr (Chapra, 1996), developed areas are likely to contribute ~8 times as much phosphorus per unit area as undeveloped areas in the watershed. Detention ponds can be reliably designed to achieve 50-60% reductions in runoff phosphorus loads from developed areas (Walker, 1987; USEPA, 1993). Temporary diversion of portions of the Platte River flow into detention area(s) in the lower watershed may also be feasible. Historically, the hatchery has occasionally operated as a treatment facility for runoff loads from the watershed (months with negative net load, Figure 7). Further studies would be needed to identify and evaluate specific alternatives to accomplish reductions in watershed loads that would offset hatchery loads, should such reductions be necessary.

Given the fact that the hatchery currently accounts for <4% of the total lake load, future lake concentrations will depend more on loads from the watershed and other sources than on loads from the hatchery, assuming that the latter are not allowed to increase. Development or other changes in the watershed could have significant impacts on the lake, regardless of hatchery discharge levels. Once acceptable discharge limits are set for the hatchery for a given permit interval, its performance should be gauged based upon whether it operates within its limits, not based upon whether lake concentrations exceed 8 ppb or any other goal.

Conclusions

1. The 1990-1996 monitoring program provided flow and concentration data for estimating phosphorus loads from the Platte River and North Branch. Phosphorus concentrations at watershed monitoring stations varied seasonally and generally increased with flow. Concentrations and loads were more responsive to precipitation events in January-July compared with August-December. The maximum response was observed in June. When adjusted for variations in flow, phosphorus concentrations in the North Branch of the Platte River and Brundage Creek exhibited decreasing trends. Total loads from the Platte River watershed ranged from 1900 to 2892 kg/yr. Stream sampling and watershed load estimates were limited by data gaps and infrequent sampling of runoff events. The range of flows and loads experienced in 1990-1996 were fairly representative of those estimated for a longer (1975-1996) period of record.

2. Estimates of phosphorus loads from the watershed and other sources are generally consistent with those developed in previous lake studies. In 1990-1992, phosphorus loads from the Platte River hatchery averaged 316 kg/yr and accounted for 9.7% of the total estimated load to Platte Lake. In 1993-1996, hatchery loads were reduced to an average of 96 kg/yr and accounted for 3.9% of the total load. Primarily as a result of hydrologic variations, watershed loads also decreased over the 1990-1996 period.

3. Lake total phosphorus loads decreased by 24% between 1990-1992 (3247 kg/yr) and 1993-1996 (2463 kg/yr) and the lake volume-weighted mean concentration decreased by 9% (from 8.5 to 7.7 ppb). The lake's response to reductions in external load was apparently buffered by phosphorus recycling from bottom sediments.

4. Trend analyses reveal significant improvements in lake water quality within the entire period of record (1976-1996) and within the recent period of intensive monitoring (1990-1996). Long-term decreasing trends in phosphorus concentration and increasing trends in transparency are believed to reflect long-term reductions in hatchery phosphorus loads. Within 1990-1996, decreasing trends in volume-weighted-mean phosphorus concentration are detected in the winter months (October-April), but not during the growing season (May-September). Similarly, increasing trends in transparency are detected in the winter months but not in the growing season. Increasing trends in April-June hypolimnetic dissolved oxygen concentrations may reflect reductions in algal productivity. Apparent trends in the 1990-1996 period are attributed to the combined effects of climatologic variations (which caused higher runoff loads in 1990, as compared with the remaining years) and reductions in hatchery phosphorus load.

5. Correlations between lake phosphorus concentrations and transparency are evident on weekly, monthly, seasonal, and annual time scales. These reflect controlling effects of phosphorus concentration on algal concentrations and photosynthesis rates. Minimum transparencies are typically measured in June and are controlled largely by calcite precipitation (whiting events). The intensities of whiting events are regulated by water temperature and pH. Photosynthesis plays a role in these events by elevating pH and reducing calcium carbonate solubility. Photosynthesis rates are controlled by a variety of factors, including nutrient concentrations, weather, water temperature, and vertical stratification.

6. Lay monitoring data reveal significant transparency increases in response to reductions in hatchery phosphorus load over the 1977-1996 period. Most of the improvements resulted from load reductions occurring prior to 1990. Between 1982-1988 and 1983-1988, the average phosphorus load from the hatchery decreased by 33% and the frequency of Secchi depths below 10 feet decreased from 59% to 27%. Between 1989-1992 and 1993-1996, the average phosphorus load from the hatchery decreased by an additional 67% and frequency of Secchi depths below 10 feet decreased from 27% to 25%. The diminishing transparency response to reductions in hatchery load reflects the fact that the percentage of the lake total phosphorus load contributed by the hatchery declined from ~34% in 1977-1981 to <4% in 1993-1996. Further improvements in transparency are projected under 1993-1996 loading regimes as recycling of historical phosphorus loads from lake bottom sediments diminishes.

7. Year-to-year variations in transparency in 1990-1996 were apparently controlled largely by physical parameters (temperature and vertical stratification) and by variations in phosphorus cycling (i.e., correlation between lower P settling rates and more intense whiting events in 1992, 1994, and 1995), rather than by variations in external phosphorus load. These factors explain the absence of a decreasing trend in May-September Secchi depths between 1990-1996, despite the substantial decreases in external load over this period.

8. The importance of phosphorus recycling in the lake is indicated by strong seasonal variations in the net settling rate, buildup phosphorus in the bottom waters during stratified periods, and by the fact that average water-column phosphorus concentrations did not respond proportionately to variations in external load. A peak in sediment phosphorus content at ~ 3 cm depth probably reflects historical phosphorus loads and indicates that there may be positive driving force for diffusion of phosphorus from the sediments into the water column. Under 1990-1996 conditions, phosphorus buildup in the hypolimnion was not correlated with loss of oxygen. Recycling mechanisms driven by iron reduction were apparently unimportant, possibly because redox potential was not sufficiently low to result in significant iron reduction and phosphorus release.

9. Hypothesis tests constructed with 1990-1996 data indicate that annual, the lake volume-weighted-mean phosphorus concentration was greater than 8 ppb in 1990 and 1992 and was less than 8 ppb in 1996. In the remaining years, the measured mean was not significantly different from 8 ppb. May-September means were significantly above 8 ppb in none of the years and were significantly below 8 ppb in 1993 and 1996. The combined annual and May-September averages for 1993-1996 (7.8 ppb and 7.5 ppb, respectively) were significantly below 8 ppb in both cases.

10. Alternative modeling approaches (monthly, annual) give similar projections of steady-state lake responses to alternative external loading scenarios. Results indicate that benefits of further reductions in hatchery phosphorus loads below 1993-1996 levels would be difficult to detect in the presence of natural seasonal and yearly variations. This largely reflects the fact that the 1993-1996 hatchery load accounted for <4% of the total lake load. Average and maximum monthly impacts of 1993-1996 hatchery loads on lake P concentrations are 0.28 and 0.48 ppb, respectively, based upon simulation of a 1975-1996 hydrologic time series.

11. Transparency measurements collected under the lay monitoring program tend to be higher than those collected under the Court Master's monitoring program by amounts ranging from 1 to 4 feet. Modeling results indicate that 1993-1996 hatchery loads are consistent with maintaining May-September transparencies in excess of 10 feet more than 90% of the time (using lay transparency as a frame of reference) and more than 68% of the time (using the Court Master's transparency data as a frame of reference). Average and maximum monthly impacts of 1993-1996 hatchery loads on May-September transparencies are –0.2 and –1.1 feet, respectively.

12. Because of their seasonal distribution, current hatchery loads have a larger impact on lake phosphorus concentrations in the late spring, as compared with other seasons. Simulations indicate that monthly-average phosphorus concentrations between April and June would exceed 8 ppb at a frequency of 21% without hatchery loads and at a frequency of 38% with hatchery loads at 1993-1996 levels. Seasonal average concentrations would be below 8 ppb in both cases. Higher lake concentrations in the spring reflect seasonal increases in watershed runoff concentrations and loads, as well as timing of hatchery loads. The net effect of spring hatchery loads is an average reduction of 0.5 to 0.6 feet in May-June, relative to base conditions of ~10 feet on the Court Master's Secchi scale and ~13 feet on the lay Secchi scale.

13. The feasibility of attaining the 8 ppb goal set by the Court (Brown, 1988ab) under any loading scenario is largely dependent upon whether the goal is interpreted as a target (central tendency for future lake measurements) or as a limit (value not to be exceeded) and upon the time scales and seasons over which the goal is

assumed to apply. Modeling and monitoring results indicate that 1993-1996 hatchery phosphorus loads are compatible with maintaining annual-average, May-September-average, and spring overturn phosphorus concentrations below 8 ppb in most years, assuming that watershed loads and other sources are constant at current levels. Excursions may occur in years with high watershed runoff.

14. If the 8 ppb goal is interpreted as a limit that applies to every sample, every day, or every month, then it is not achievable under any hatchery loading scenario without reducing phosphorus loads from other sources. Such an interpretation would be unrealistic in the context of natural variations typically found in lakes.

15. Consideration should be given to revising the current hatchery discharge permit to assure that future loads do not increase above 1993-1996 levels. This could be accomplished by a setting limit of 55 kg for any 3-month period and 133 kg for any 12-month period. Assuming that loads from sources other the hatchery do not increase, this measure should be sufficient to maintain annual and growing-season-average lake P concentrations below 8 ppb in most years. Based upon estimates of phosphorus loads prior to the hatchery expansion in 1970 ranging from 105 to 511 kg/yr, operating within a limit of 133 kg would reduce impacts to levels below those that occurred prior to the expansion.

16. If interpretations of the goal dictate that further reductions in hatchery phosphorus loads are necessary, then alternative methods for accomplishing this should be identified and evaluated from a cost-effectiveness standpoint. With further refinements in hatchery operations, it may be possible to reduce net hatchery loads below 1993-1996 levels without substantial capital costs. Additional alternatives would include further modifications to the hatchery design and/or implementation of watershed nonpoint source controls to offset hatchery loads.

17. Given the fact that the hatchery currently accounts for <4% of the total lake load, future lake concentrations will depend more on loads from the watershed and other sources than on loads from the hatchery, assuming that the latter are not allowed to increase. Once acceptable discharge limits are set for the hatchery for a given permit interval, its performance should be gauged based upon whether it operates within these limits, not based upon whether lake concentrations exceed 8 ppb or any other goal.

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A B S T R A C T

Platte Lake is a 10.6 km^2 natural impoundment located on the Platte River in Benzie County, Michigan. The lake has been classified as "oligo-mesotrophic", with relatively low nutrient concentrations, low algal productivity, depletion of dissolved oxygen in bottom waters during the summer, and alkaline chemistry. Legal proceedings were initiated in the 1980's because of concerns about the impacts of discharges from a state-operated fish hatchery on nutrient concentrations, water transparency, and related water-quality conditions. As a result of these proceedings, an 8-ppb total phosphorus goal was established by Court and the State was ordered to alter the operation of the hatchery in a manner consistent with achieving that goal. With implementation of control measures, phosphorus load from hatchery was reduced from ~33% of the total lake load in 1976-1981 to <4% in 1993-1996.

This report analyzes monitoring data collected under intensive lake and watershed surveys between 1990 and 1996. Monitoring data from 1976 to 1989 provide an historical frame of reference. Long-term responses to historical reductions in hatchery phosphorus load are quantified through data analysis and modeling. Current and projected future conditions are evaluated with particular reference to the 8-ppb goal and to a 10-foot transparency benchmark identified in legal proceedings.

Trend analyses reveal significant decreases in lake phosphorus concentrations and increases in transparency following historical reductions in hatchery load. Most of the potential benefits of hatchery control were achieved by load reductions which occurred prior to 1990. Year-to-year variations in transparency between 1990-1996 were controlled largely by climatologic factors (influencing water temperature and vertical stratification) and by variations in the cycling of phosphorus within the lake, rather than by variations in external phosphorus load. Further improvements are projected under 1993-1996 loading regimes, as recycling of historical phosphorus loads from lake bottom sediments diminishes.

Monitoring and modeling results indicate that 1993-1996 hatchery loads are compatible with maintaining annual-average, growing-season average, and spring overturn phosphorus concentrations below 8 ppb in most years. Excursions may occur in years with high watershed runoff. If the 8-ppb goal is interpreted as a limit that applies to every sample, every day, or every month, then it is not achievable under any hatchery loading scenario without reducing phosphorus loads from other sources. In the context of long-term lay monitoring data, 1993-1996 hatchery loads are compatible with maintaining transparencies in excess of 10 feet more than 90% of the time. This is in sharp contrast to conditions observed during the period of peak hatchery loading (1977-1981), when transparencies exceeded 10 feet only 4% of the time.

Recommendations are made for revising the hatchery discharge permit to ensure that future loads do not increase above 1993-1996 levels. Assuming that loads from sources other than the hatchery do not increase, this measure should be compatible with lake conditions described above. If interpretations of the 8-ppb goal dictate that further reductions in hatchery phosphorus load are necessary, then alternatives for accomplishing these reductions should be identified and evaluated from a cost-effectiveness standpoint. Additional alternatives would include further refinements to hatchery operations, modifications to hatchery design, and/or implementation of watershed nonpoint source controls to offset hatchery loads.